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WATER QUALITY IMPACTS OF URBANISATION

EVALUATION OF CURRENT RESEARCH

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ABSTRACT

This report is the first in a series of reports focussing on the water quality impacts of urbanisation. The primary objectives of this report has been to critically review relevant published research, to identify important areas where there is a current lack of in-depth knowledge and to define future research directions.

It is common knowledge that urbanisation can lead to significant water quantity and quality impacts. Past research into quantity impacts have resulted in an in-depth understanding of these issues and acceptable reliability in commonly available predictive approaches. However this is not the case for water quality impacts. The underlying processes and concepts relating to urban water quality are well known in a qualitative sense. However their quantification has proved to be extremely difficult. This is a major failure in most research studies. As a result, attempts to correlate land use to pollutant loadings have been inconclusive.

A limitation in current urban water quality research is that the approaches adopted are strongly based in water quantity research undertaken in the past. The extension of these concepts and processes is not satisfactory due to the strong reliance on physical factors only and the limited recognition of chemical processes. Chemical processes exert a strong influence on urban stormwater quality characteristics. It is this neglect which can be primarily attributed to the often contradictory results reported in research studies and the strongly location specific nature of study outcomes. As such, this has led to significant constraints in defining the process kinetics of pollutant generation, transmission and dispersion such as pollutant build-up and wash-off.

Consequently, the management of water quality impacts in urban areas has proven to be a difficult task. The effectiveness of commonly adopted management and structural measures is open to question. The contradictory research findings in relation to these measures clearly point to the significant role played by location specific factors influencing water quality rather than purely land use.

A holistic approach is needed to safeguard the quality of receiving waters in urban areas. The current approach to urban water quality management is piecemeal and the benefits are only be marginal. It provides a false sense of achievement and even detracts attention from the more difficult challenges to be met to safeguard urban water quality.

It is important to ensure the transferability of research outcomes for wider benefit and the relationships derived should facilitate this transfer. Future research directions have been proposed taking the above noted concerns into consideration.

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LIST OF ABBREVIATIONS

Al	Aluminium
BOD	Biochemical oxygen demand
Cd	Cadmium
COD	Chemical oxygen demand
Cr	Chromium
Cu	Copper
DOC	Dissolved organic carbon
Fe	Iron
Hg	Mercury
Mn	Manganese
NH_3	Ammonia
Ni	Nickel
NO_2	Nitrite
NO_3	Nitrate
PAH	Polycyclic aromatic hydrocarbons
Pb	Lead
SS	Suspended solids
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
V	Vanadium
Zn	Zinc

WATER QUALITY IMPACTS OF URBANISATION

EVALUATION OF CURRENT RESEARCH

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1. INTRODUCTION

The spread of urbanisation is a common phenomenon witnessed in most parts of the world. Population growth together with industrialisation, wealth creation and improved mobility have resulted in the irrevocable transformation of previously rural land into housing developments and the more intensive development of urban fringe areas. As this urban fabric is formed, it gives rise to a host of environmental impacts. Therefore it is imperative that this incessant urban growth is astutely managed and innovative strategies are adopted to ensure the protection of key environmental values in a region. The appropriate and prudent management of urbanisation impacts pose significant challenges to regulatory organisations.

Urban expansion transforms local environments and can dramatically alter local conditions and in particular the rate of movement of pollutants into waterways, thereby adversely changing the quality of water. In the context of effective urban resource planning and management, the recognition of the impacts of urbanisation on the water environment is among the most crucial. The significance stems from the fact that water environments are greatly valued in urban areas as environmental, aesthetic and recreational assets. Arguably, it is the water environment which is most adversely affected by urbanisation. Any type of activity in a catchment that changes the existing land use will have a direct impact on its hydrologic regime and water quality characteristics. The deterioration of water quality, degradation of stream habitats, and flooding, are among the most tangible of the resulting detrimental impacts. These consequences are due to the removal of vegetation and the replacement of previously pervious areas with impervious surfaces and the introduction of pollutants of physical, chemical and biological origin, resulting from various anthropogenic activities. Therefore the appropriate management of urban stormwater runoff and streamflow has significant socio-economic and environmental ramifications.

In an effort to mitigate the adverse impacts of urbanisation, the current approach by authorities is the adoption of structural and/or regulatory measures. Structural measures commonly include the provision of detention basins, wetlands and gross pollutant and sediment traps. Regulatory measures are often in the form of restrictive zoning, demarcation of buffer strips and the imposition of limits on stormwater quantity and quality exports from an urban development. Quite often structural measures are adopted as a result of regulatory requirements.

However for these measures to be effective, the availability of predictive methodology for the reliable assessment of urbanisation impacts on the water environment is essential. This in turn requires an in-depth understanding of the concepts and processes which influence urban water quality. Analysis of 'what if' scenarios for evaluation of land development alternatives and the development of compensatory strategies will be greatly enhanced with the availability of more reliable assessment capabilities and greater understanding of the inherent ecological factors associated with urbanisation.

2. REPORT DETAILS

2.1 Background to the Report

This document is the first in a series of reports focussing on water quality impacts of urbanisation. The research project was initiated at the request of the Gold Coast City Council and the Built Environment Research Unit of the Department of Public Works. The major objective of the research undertaken is to relate stormwater runoff quality to different urban forms. This report consists of a 'state of the art' review of research undertaken in the arena of urban water quality. In keeping with the principal objective of the project, the report focuses on important concepts and processes in relation to urban water quality and primary water pollutants. This report acted as a springboard for the ongoing experimental study and analysis being undertaken as part of the project.

2.2 Report Objectives

The safeguarding of urban water quality is being afforded increasing importance due to the recognition of urban stormwater resources being important environmental assets. It is in this context that the design of the urban form is being subjected to greater scrutiny and innovations adopted in order to minimise its ecological footprint in relation to the water environment. However the relationships between urban form and water quality are not intuitively obvious. This is because the underlying processes which influence pollutant generation, transmission and dispersion are complex and poorly understood. These processes do not lend themselves to simple mathematical modelling.

The key role played by various anthropogenic activities and the difficulty in their mathematical formulation further adds to the complexity of the inherent processes. Consequently, the mere adoption of structural or regulatory measures for urban water pollution mitigation will not suffice. It is important that the mitigative management strategies adopted are appropriately formulated based on a comprehensive awareness of influential factors. This requires a multi faceted strategy that would encompass:

- The continuous improvement and/or development of strategies based on currently available ‘state of the art’ research outcomes.
- The undertaking of practical research in areas where there is a discernible lack of in-depth knowledge.

This ‘state of the art’ evaluation of research brings together significant work undertaken locally and overseas. It has been undertaken with a two-fold objective. Firstly, it is to critically review relevant research outcomes in the arena of urban water quality. Secondly, it is to identify important areas where the current knowledge is inadequate. In the long term it is hoped that this report will implicitly contribute to the development of a comprehensive knowledge base, which will form the basis for the formulation of credible and innovative urban growth management strategies to mitigate the adverse environmental impacts commonly associated with urbanisation.

2.3 Scope and Outline of the Report

This research review is based on published research outcomes and focuses on important concepts and processes governing urban stormwater quality and primary water pollutants of physical and chemical origin. The microbiological quality of water did not form a part of this review.

Chapter 3 of the report discusses the impacts of urbanisation on catchments. These impacts include significant modifications to the hydrologic regime and stormwater runoff quality. The discussion encompasses the sources of pollutants, common pollutant pathways and the fundamental concepts inherent in pollutant build-up and wash-off. The primary water pollutants are discussed in Chapter 4. The pollutants discussed include, litter, suspended solids, plant nutrients, heavy metals, hydrocarbons and organic carbon. The impact of these pollutants on the environment, their sources and pathways and the primary factors which influence these factors are covered in this discussion.

The correlation of land use with pollutant loadings is the topic in Chapter 5. This is a common focus in numerous research studies. Its importance stems from the fact that a thorough understanding of these issues will contribute significantly to the development of urban planning policies and the adoption appropriate mitigative management strategies. Chapter 6 has been devoted to the discussion of the current state of knowledge in relation urban water quality. It brings together the important outcomes from various research studies including a comprehensive summary of the conclusions from the review undertaken. It also includes a discussion on management implications and provides directions for future research to be undertaken in the urban water quality arena. Chapter 7 provides brief conclusions of the outcomes of the review.

3. IMPACTS OF URBANISATION

Land use modifications associated with urbanisation are invariably reflected in the stream flow regime. This is mainly as a result of changes to the characteristics of the surface runoff hydrograph. Additionally, the water quality characteristics can undergo significant changes. The quality and quantity impacts of urbanisation have been well documented in research literature (for example ASCE 1975; Codner et al. 1988; Hall & Ellis 1985; House et al. 1993; Mein & Goyen 1988).

3.1 HYDROLOGIC REGIME

The following review discusses the quantity impacts on surface runoff due to catchment urbanisation. The quantity impacts of urbanisation can be directly attributed to the physiographic changes to the catchment. These changes include:

1. Removal of vegetation which results in:
   - reduced evapotranspiration losses
   - reduced surface roughness and catchment storage

2. Increase in impervious area which results in:
   - reduced infiltration losses
   - reduced depression storage
   - more uniform surface slopes

1. Drainage channel modifications which result in increased hydraulic conveyance efficiency.

Consequently the hydrologic behaviour of a catchment and in turn the streamflow regime undergoes significant changes. These changes are apparent not only during a rainfall event, but also during dry periods. The number of runoff events tends to increase relative to a rural catchment, with even relatively low rainfall intensity events producing runoff (Codner et al. 1988). Instances have been cited by Hollis (1975) and Waananen (1969) where previously ephemeral streams have become perennial with catchment urbanisation. Crippen (1965) has noted that the hydrologic changes observed are due to the composite effect of various catchment modifications. It is generally difficult to ascribe a specific hydrologic change to a particular detail of catchment alteration. The hydrologic changes that urban catchments commonly exhibit are:

- increased runoff hydrograph peak;
- increased runoff volume;
- reduced time of concentration and catchment lag;
- reduced catchment and channel storage;
- changed base flow conditions.

(ASCE 1975; Bedient et al. 1985; Cordery 1976; Codner et al. 1988; Delleur 1982; Waananen 1969).

3.1.1 Runoff Hydrograph

Urbanisation makes a catchment 'flashy' (Mein & Goyen 1988). As illustrated in Figure 1 below, the runoff hydrograph shape can undergo considerable changes due to urbanisation. The most obvious of these changes is the sharp rise in the peak flow and the reduced time base of the hydrograph. Rao and Delleur (1974) have shown that the modification of peak discharge is the most important impact of urbanisation.

The rise in hydrograph peak with increasing urbanisation can be attributed to two primary mechanisms. The first is the replacement of vegetation with impervious surfaces of relatively less roughness and uniform slope. The other is the provision of gutters, road drains and storm sewers of low roughness in lieu of the natural channels. These factors combine to reduce the time of concentration and catchment lag (McPherson 1974; Codner et al. 1988), with a resulting increase in the runoff peak. Waananen (1969) has shown that the lag can reduce by as much as 70% for an urban

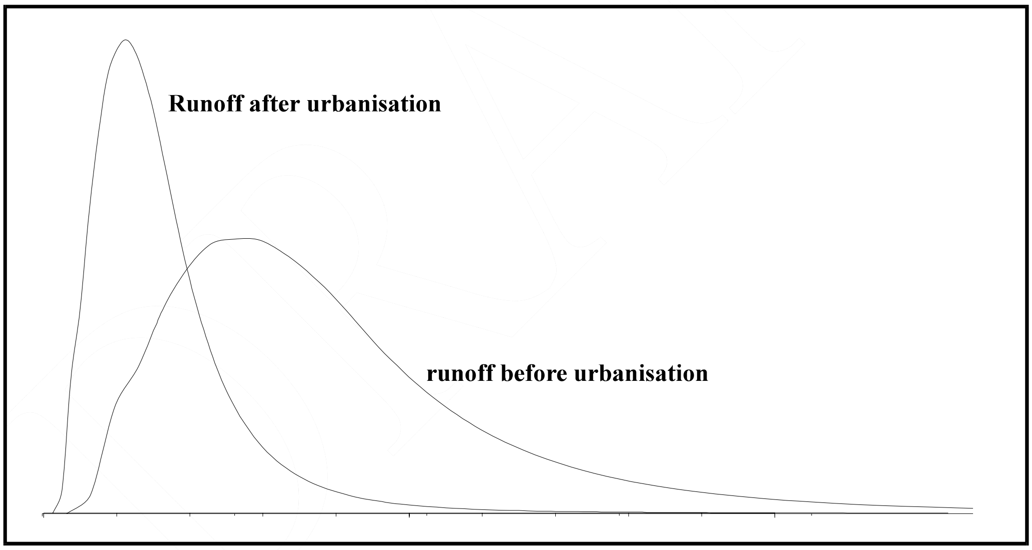

Figure 1 - Runoff hydrographs from similar urban and rural catchments

catchment. Cech and Assaf (1976) found that urbanisation is highly significant for more frequent flood events. Its significance reduces in importance for less frequent events. Hollis (1975) found that flood peaks for storms in the range of 150 to 200 years ARI are not affected by urbanisation. The reasons attributed for the reducing influence of urbanisation with increasing storm recurrence interval are:

- High intensity rainfall bursts which are generally identified with storms of high ARI are often preceded by lower intensity rainfall. This rainfall in most instances is adequate to meet the soil moisture requirements of the catchment. Therefore during the subsequent high intensity rainfall burst, the infiltration losses will be sufficiently low to make the catchment surface behave in an almost impermeable manner (Hollis 1975).
- During low intensity rainfall, the losses and surface flow velocities will be significantly different for pervious and impervious surfaces. However, for higher intensity events which are generally associated with high recurrence intervals, large quantities of water will be present on the surface. As such, the losses and surface roughness will not be significant and the flow velocity over different surface types will not be greatly different (Boyd et al. 1987; Espey & Winslow 1974).
- There could be some 'throttling of flow' in drains and sewers during severe storms (Wilson 1967).

The crucial factors responsible for the increase in peak runoff with urbanisation include the type and extent of impervious cover, the layout of the drainage network and the spatial distribution of urban areas. As an example, if the urbanised area is located very close to the catchment outlet, the rapid runoff from this area could reach the outlet prior to the contributions from the other areas. This would result in a more attenuated runoff hydrograph. Bonuccelli and Hartigan (1978) found that for the catchment investigated by them, the location of the urban area in the middle and upper middle third of the catchment is the most sensitive to increases in urbanisation.

Goonetilleke and Jenkins (1999) have shown that in a catchment with more than one major stream tributary, if the urbanisation is mostly confined to one tributary, it could lead to attenuation of the hydrograph at the main outlet. Also the built-up area could consist of a substantial amount of pervious areas provided by lawns and recreational areas. This would result in a relatively reduced impact when compared to an industrial or commercial area with the same extent of built-up area (Bhaskar 1988; Brater & Sangal 1969; Carroll 1995; Packman 1980; Waananen 1969; Wong & Chen 1993). The above examples illustrate the fact that a catchment does not necessarily behave in an intuitively obvious manner with urbanisation.

Investigators have reported increases in peak runoff between 1.3 to 6 times their value under rural conditions due to urbanisation (Espey et al. 1969; James 1965; Sawyer 1963; Seaburn 1969; Tholin & Keifer 1960; Waananen 1961; Wilson 1967). A generalised relationship between percentage increase in peak runoff to percentage urbanised for different ARI values has been derived by Hollis (1975). He has used data given by a number of other researchers as shown in Figure 2. The validity of the relationship thus derived is however questionable, as the increase in peak runoff is also dependent on a number of other catchment physiographic variables such as:

- the soil conditions;
- the stream network layout;
- the stream channel improvements; and
- the degree of imperviousness of the urbanised area.

Hollis (1975) also agrees that additional catchment variables should be included to improve the prediction validity of the derived relationship.

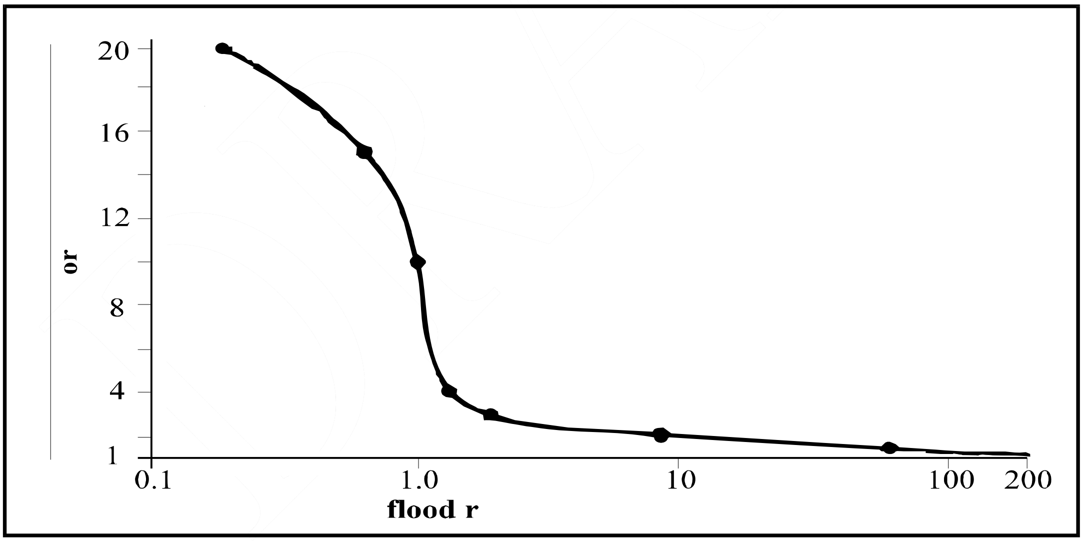

Figure 2 - Comparison of peak runoff with percentage urbanised (adapted from Hollis 1975)

3.1.2 Runoff Volume

Urbanisation leads to an increase in runoff volume as well as annual yield (ASCE 1975; Sawyer 1963; Seaburn 1969; Wilson 1967). In the study of a catchment in Mississippi, Wilson (1967) concluded that the mean annual flood volume would increase by a factor of 4.5 under total urbanisation. Seaburn (1969) reports increases in the range of 1.1 to 4.6 times the storm runoff volume when compared with rural conditions, depending on the ARI of the individual storm. However, Ferguson and Suckling (1990) in their study of a 222 km^2 partially urbanised catchment in Georgia found that the total annual runoff volume increased during ‘wet’ years, but decreased during ‘dry’ years. The reasons attributed by them for these observations, which are not in complete agreement with other studies are:

- The evapotranspiration in urban areas is not reduced in proportion to the loss of vegetated areas. In urban areas, evapotranspiration losses are increased by exposure of the vegetation to advection of heat from surrounding surfaces. Therefore the increased evapotranspiration from remaining vegetation can be partially compensating.
- Though this phenomenon of increased evapotranspiration is always present, its impact on runoff is masked during the ‘wet’ years.
- The increased evapotranspiration is further assisted during ‘dry’ years due to clear skies, high solar radiation, high temperature and low humidity in a tropical climate.
- In most urbanised catchments, the drainage reservations and wetlands generally retain their vegetation. It is this vegetation that is most effective in evapotranspiration.

However, the results reported by James (1965) from a long-term continuous simulation study using the Stanford Watershed Model on a 186 km^2 partially urbanised catchment in California, is at variance with the results reported by Ferguson and Suckling (1990). He found that surface runoff volume increased by almost six times its rural value for the wettest year to 125 times for the driest year. Considering the conflicting conclusions derived from various research studies, it can be postulated that catchment characteristics and also the climate play a key role in the magnitude of increase in runoff volume and yield.

3.1.3 Base Flow

Contradictory results have been reported in research literature with regards to changes in base flow with urbanisation (ASCE 1975; Codner et al. 1988; James 1965; Sawyer 1963; Waananen 1969). However, it is possible to explain the reasons for the inconsistency in reported results. There is no doubt that the paving of previously pervious surfaces leads to reduced groundwater recharge. However the presence of detention basins would greatly increase infiltration. Additionally, the importation of water for lawn sprinkling in residential areas and leakage from sewers and water supply pipelines would further add to the groundwater recharge. Ferguson and Suckling (1990) also note that the location of vegetation would impact on low flows. Removing vegetation from upland areas may cause only a small reduction in the total evapotranspiration loss in ground water available as base flow. Vegetation in floodplains, along channels and in wetlands has higher rates of evapotranspiration and will continue to contribute to losses and thereby reduce the base flow. Therefore base flow changes are dependent on catchment characteristics.

3.2 STORMWATER QUALITY

Urbanisation not only impacts on the hydrologic regime of catchments, but also has a profound influence on the quality of stormwater runoff. Consequently, urbanisation will also alter water quality in receiving waters. Rainfall and the resulting surface runoff washes and cleanses the air and the land surface, and then transports a variety of materials of chemical and biological origin to the nearest receiving water body. These contaminants will detrimentally impact on aquatic organisms and alter the characteristics of the ecosystem. This results in a water body which is fundamentally changed from its natural state (House et al. 1993).

The changes to the hydrologic regimes of urban areas such as the increase in stormwater runoff velocities and volumes will lead to enhanced erosion, dislodgement, entrainment and solubility of pollutants present on the catchment surface (Simpson & Stone 1988). Sonzogni et al. (1980) in their study found that suspended solids and nutrients from urban areas ranged from 10 to 100 times greater than loads from equivalent undisturbed land. Similar observations relating to increases in nutrient loads and other pollutants have been reported by numerous researchers (for example Line et al. 2002; Lopes et al. 1995; Meister & Kefer 1981; Owens & Walling 2002; Wahl et al. 1997).

As Ahyerre et al. (1998) have noted, the generation and transport of pollutants in urban systems during a storm event is very complex as it concerns many media, many space and time scales. The urban environment is affected by a variety of anthropogenic activities. Roads, housing, commerce and industry not only lead irrevocable changes to the urban landscape, but are also responsible for introducing numerous pollutants to the environment. The major problems in urban areas are, the pollution of the atmosphere, soil and water. As an example, Lind and Karro (1995) found that heavy metal concentrations in the topsoil layers of urban roadside areas in Sweden to be 2 to 8 times higher when compared to rural areas.

Urban stormwater runoff has been recognised as a major source of a wide variety of pollutants to water bodies. Recent years have witnessed significant advances in the control of point sources of pollution such as sewage outfalls. Consequently, non-point sources such as stormwater runoff are gaining increasing importance (Bedient et al. 1980; Bradford 1977). The pollutant impact associated with stormwater runoff in terms of concentration and total load can be significantly higher than secondary treated domestic sewage effluent (Droste & Hartt 1975; Helsel et al. 1979; Wanielista et al. 1977; Yu et al. 1975). This applies not only to the physical and chemical quality, but also to the microbiological quality of urban stormwater (Qureshi & Dutka 1979). As Pitt (1979) has noted, stormwater runoff treatment may be a more effective water quality control measure than further improvements in wastewater effluent.

During a rainfall event, the impacts of high flows and intermittent discharges of pollutants on receiving water bodies are superimposed on the hydrologic, physico-chemical and biological characteristics of an urban catchment. Urban stormwater runoff will produce both, short-term and long-term changes in receiving waters leading to habitat instability and chemical toxicity. This in turn will result in changes to aquatic communities such as increased mortality of biota and detrimental changes to species diversity and abundance (House et al. 1993; Lopes & Fossum 1995; Whal et al. 1997). These changes will reflect the influence of urban runoff characteristics and not the natural variability of environmental conditions. Consequently, the combination of changes to the physical habitat and altered water quality is the major impact of urban stormwater runoff (Collier et al. 1998; Field & Pitt 1990; House et al. 1993; Warren et al. 2003). Therefore though stormwater runoff events are episodic what is of serious concern is the shock pollutant load on receiving waters resulting from a stormwater runoff event (Bradford 1977; Cordery 1977; Overton & Meadows 1976; Pitt 1979).

3.2.1 Pollutant Sources

As the stormwater flows over the drained surface, pollutants will be incorporated through various physical and chemical processes (Mikkelsen et al. 1994). The source from which the stormwater runoff is derived is one of the most important factors which will influence its pollutant composition. The primary pollutant sources in an urban catchment are:

- street surfaces
- industrial processes
- construction and demolition activities
- corrosion of materials
- vegetation input
- litter
- spills
- erosion

(Pitt 1979; Pitt et al. 1995).

A. Street surfaces

Street surfaces and by implication vehicular traffic is the single most important source of urban water pollution (Bannerman et al. 1993; Sartor & Boyd 1972). These two factors need to be considered in conjunction as they act synergistically to contribute to urban stormwater pollution.

Streets have a profound impact on stormwater runoff quality as they constitute a high percentage of impervious surfaces in an urban area. Also most importantly, streets provide an efficient stormwater conveyance system to receiving waters during a rainfall event. Materials present in street surfaces can originate from a range of sources such as:

- street surface degradation
- vehicle lubrication system losses
- vehicle exhaust emissions
- load losses from vehicles
- degradation of vehicle tyres and brake linings
- particulate materials from local soils

(Brinkmann 1985; Goettle & Krauth 1980; Shaheen 1975).

Sartor and Boyd (1972) undertook a comprehensive study into street surface contaminants. They found that the quantity of contaminant material on street surfaces vary widely, depending on a range of factors. This included the length of time which had elapsed since the street was last cleaned either by sweeping or rainfall flushing, surrounding land use, traffic volume and other traffic characteristics, street surface characteristics and maintenance practices. They have provided the following pollutant loading rates for different land use areas as given in Table 1.

Table 1 – Street surface pollutant loading rates for different land uses (adapted from Sartor & Boyd 1972)

Land use	Loading rate (T/km)
Commercial	0.08
Residential	0.34
Industrial	0.80

Sartor and Boyd (1972) have further attributed the high pollutant loading rate in industrial areas to reasons such as less frequent sweeping, spillage from vehicles and streets being in poor condition. In contrast, the reason for commercial areas having the lowest pollutant loading rate was attributed to more frequent street sweeping. However it is important to note that these values are for a study undertaken in the United States and may not be transferable to other geographical regions due to climatic, anthropogenic and technical factors.

Vehicle traffic contributes solid, liquid and gaseous pollutants. Abrasion products from tyres and brake linings would depend on traffic volume, road characteristics such as the location of traffic lights, road layout, pavement surface and driver habits. The wear of the road pavement would depend on its condition, maintenance practices, weather conditions and traffic volume. Spillage of fuel, oil and lubricants are found everywhere on roads, but they are generally concentrated in car parks and near traffic lights. The gaseous products would initially contribute to atmospheric pollution but would eventually return to the ground due to wet deposition during rainfall and thereby contribute to stormwater runoff pollution (Brinkmann 1985; Mikkelsen et al. 1994; Novotny et al. 1985). The pollution generated by vehicles is mostly confined to the street surface. Hewitt and Rashed (1990) found that heavy metals and hydrocarbons emitted by vehicles are deposited within 50m of the carriageway. A very rapid decline in pollutant deposition fluxes with distance from the road centre was observed and the impact of vehicles was found to be restricted to a narrow band on either side of a roadway.

The study undertaken by Van Metre et al. (2000) clearly illustrates the important role played by vehicle traffic in stormwater pollution. They found that increases in hydrocarbon concentrations in a number of water bodies could not be attributed solely to urbanisation. Concentrations had also increased in catchments where urbanisation was stable, but where there was an increase in automobile usage. These conclusions are also supported by Larkin and Hall (1998) who found that the hydrocarbon concentration in stormwater runoff from roads corresponded closely with local traffic conditions.

According to Sartor and Boyd (1972), the street surface characteristics which were found to have an impact on pollutant loadings include pavement material and pavement conditions. Asphalt pavements were found to have 80% more loading than concrete roads. Similarly streets in fair-to-poor condition had loadings about 250% higher than streets in good-to-fair condition. However Shaheen (1975) in his study failed to detect any discernable impact on the build-up of street surface contaminants due to factors such as speed, traffic mix or the composition of the road paving material. A number of reasons can be attributed to these contradictory results such as sampling design, measurement errors and location specific factors.

B. Industrial processes

Industrial processes are an important source for a range of pollutants in an urban area. The nature and concentration of pollutants in stormwater from industrial sites is dependent on the nature of the industry and the management of the facility. Stacks, fugitive emissions and spills are the causes of pollutant releases to the environment. The relatively high pollutant concentrations in stormwater runoff from industrial sites, when compared to other land uses have been noted by many researchers (for example Fam et al. 1987; Kelly et al. 1996; Sartor & Boyd 1972).

C. Construction and demolition activities

Construction and demolition debris have the potential to contribute significant quantities of sediments and litter to the urban environment. The quantities would essentially depend on the management of the site, its extent and erosion control measures in place. Line et al. (2002) in an evaluation of pollutant export from a range of land uses in the United States found that the sediment export rate was more than 10 times the value for construction sites when compared to other land uses, whilst Konno and Nonomura (1981) reported that the sediment load can be as high as 100 times based on a study undertaken in Japan.

D. Corrosion of materials

Acid rain and aggressive gases can produce appreciable corrosion of roofs, gutters and other metal surfaces. Corrosion rates will depend on the availability of corrodible materials, the frequency and intensity of exposure to an aggressive environment, the drying-wetting frequency of the exposed surfaces, the character and structure of the materials and maintenance practices (Brinkmann 1985). In regions where metallic roofs are common, corrosion can be a significant source of stormwater pollution. As examples, studies undertaken by Bannerman et al. (1993), Gromaire-Mertz et al. (1999) and Quek and Forster (1993) found that heavy metal concentrations in runoff from galvanized roofs was higher when compared to runoff from streets.

Similar conclusions were derived by Davis et al. (2001) in a comprehensive study of urban pollutant sources. Their study included a variety of building sides and roofs among other surfaces. Results obtained indicated that brick and painted wooden buildings were responsible for relatively high metal concentrations. Also based on the clear differences between different types of buildings, it was evident that the building material itself was the pollutant source, and not that the buildings were collecting atmospheric deposits.

E. Vegetation input

This includes leaves and other plant materials such as pollen, bark, twigs and grass. Vegetation input can be a significant source of nutrients in urban areas with high canopy cover. Novotny et al. (1985) reported that a mature tree can produce between 15 to 25kg of leaf residue during the fall season. Though the actual input rate would be dependent on the season, climatic conditions, land use, local landscaping and public works practices, vegetation fragments can be quite widespread in urban pollutants. However Allison et al. (1998) have questioned the importance of leaf litter as a nutrient source. Based on the outcomes of a study on an urban area in inner-city Melbourne, they found that the leaf litter was about two orders of magnitude smaller than the nutrient loads measured in stormwater samples. These observations confirm the very significant role played by location specific factors in dictating the characteristics of urban stormwater quality.

F. Spills

This category of contaminant is difficult to define quantitatively, either in terms of volume or composition or even to predict its occurrence. The major source of spills is vehicular transport. The types of materials vary widely and generally include building and landscaping materials, bulk commercial and industrial raw materials and various types of wastes (Sartor & Boyd 1972).

G. Erosion

This particularly refers to the erosion of stream banks, pervious surfaces and material stockpiles at construction, demolition, industrial, commercial and waste disposal sites (Nelson & Booth 2002; Novotny et al. 1985; Wahl et al. 1997). Stream banks are particularly prone to increased erosion due to changes to the hydrologic regime which could lead to higher peak flows. Nelson and Booth (2002) in a study of a 144 km^2 urbanising catchment found that the annual sediment yield had increased by nearly 50% due to urban development with stream bank erosion accounting for about 20%.

3.2.2 Pollutant Pathways

The primary pollutant pathways are:

- wet and dry atmospheric deposition;
- wash-off of contaminants deposited on the ground and other surfaces;

(Goettle & Krauth 1980; Novotny et al. 1985).

A. Wet and dry atmospheric deposition

Wet and dry atmospheric deposition essentially relates to the transmission of pollution through dustfall, rainfall, mist and fog. These processes are important contributors to the total catchment pollutant load and stormwater runoff quality. The atmospheric contaminants are present in the form of solid, liquid and gaseous substances and are washed out by rain or mist or deposited as sediments. Common substances include, carbon monoxide, sulfer dioxide, nitrogen oxides, hydrocarbons and dust. They are brought into the urban atmosphere from long distances or could be emitted from various sources either on a regional or local scale. Some of the atmospheric pollutants in the solute or gaseous phases will undergo further synthesis due to physical, chemical or photochemical processes (Brinkmann 1985; Fenger 1999; Goettle & Krauth 1980; Novotny et al. 1985; Novotny & Goodrich-Mahoney 1978).

The concentration of contaminants in the atmosphere is influenced by meteorological, topographical and land use factors. The chemical composition and concentration patterns of atmospheric pollution can vary widely, either temporally or spatially due to meteorological factors which will result in processes such as dispersion and re-suspension (Brinkmann 1985; Ebbert & Wagner 1987). Deletic et al. (1997) found that there was no correlation between dust fall-out and measured mass of accumulated solids on a paved surface. Unfortunately interrelationships between the various influential factors are poorly understood. Therefore this adds a significant uncertainty to the accurate prediction of atmospheric pollution and in turn the contribution of atmospheric sources to stormwater pollution (Namdeo et al. 2000).

Among the atmospheric contaminants, suspended and settleable solids or dust is the most obvious. Novotny et al. (1985) have defined dust as particles less than 60μm in diameter. Furthermore, they have quoted a dust deposition rate of $50mg/m^2$.day for atmospheric fallout with particle sizes <30µm making up 99% of the total. However the applicable geographical area or the land use has not been mentioned. In contrast, Goettle and Krauth (1980) have given dustfall rates for Germany for different land uses ranging from rural to industrial areas. The rates given range from 85mg/m^2.day to 2000mg/m^2.day. It is not possible to verify the reasons for the very wide variation in values in the two studies. However it was noted that Goettle and Krauth (1980) have not given a particle size definition for dust. Also, as House et al. (1993) have noted, the differences in quoted values for various pollutants highlights the common problem of characterising urban pollution and stormwater runoff quality.

The washout of atmospheric pollutants due to rainfall or mist is the source of wet deposition. Rainfall is a significant source of some pollutants in urban stormwater runoff. The study by Ebbert and Wagner (1987) found that the median contribution of rainfall to stormwater runoff loads of twelve constituents from thirty one urban catchments representing eight geographic locations within United States, ranged from 2% for suspended solids to 74% for total nitrogen. Novotny et al. (1985) noted that the rainfall pH in the urban areas of Wisconsin, USA was about one unit less than in regional areas. This higher acidity would have an appreciable impact on stormwater runoff quality characteristics. The overall impact of rainfall pollution superimposed on stormwater runoff can best be judged in comparison with average concentration and annual loadings. Table 2 below is reproduced from Goettle and Krauth (1985) which gives results from investigations in Munich and Zurich. In both areas, the predominant land use was residential with separate storm sewer systems. The data given highlights the polluted nature of urban rainfall. However it is important to note that these values could change significantly on a regional basis due to the location specific nature of atmospheric pollution and the influence of meteorological factors.

The washout of atmospheric pollutants by rainfall droplets is quite effective and takes place in the early stages of the rainfall. This causes a 'first flush effect' with the initial portion of the rainfall being more polluted than the remainder. Consequently, the pollutant load in the surface runoff at the early stages of a rainfall event could tend to be independent of precipitation magnitude and intensity (Hall & Ellis 1985; Novotny et al. 1985; Randall et al. 1981).

Table 2 – Pollutant wash-off from urban areas (in Munich and Zurich) (adapted from Goettle & Krauth 1985)

Pollutant	Concentrations (mg/L)		Runoff loads (kg/ha.annum)	
	Rain	**Runoff**	**Rain**	**Runoff**
Ammonia (NH_3)	1.3	0.9	3.6	2.5
Nitrite (NO_2)	0.02	0.1	0.06	0.3
Nitrate (NO_3)	2.9	2.8	8.2	7.8
Total phosphorus (TP)	0.3	0.7	0.8	1.9
Suspended solids (SS)	54	125	151	350
Chemical oxygen demand (COD)	37	52	103	145
Chrominium (Cr)	0.002	0.004	0.006	0.021
Zinc (Zn)	0.08	0.13	0.23	0.36
Copper (Cu)	0.012	0.01	0.04	0.03
Cadmium (Cd)	0.001	0.001	0.004	0.003
Lead (Pb)	0.11	0.11	0.31	0.31

However as Lewis (1981) has shown, in regions with a well-defined seasonality of rainfall, a significant proportion of the total annual atmospheric loading may be flushed out in the first few days after the onset of rain. Hence the subsequent rainfall events would be relatively less polluted. This further underlines the difficulties in characterising urban stormwater quality.

B. Wash-off of pollutants

Besides the washout of atmospheric pollutants during a rainfall event, the wash-off of pollutants built up on the ground and other surfaces is an important contributor to the surface runoff pollution. During low rainfall events, the pollutants incorporated into stormwater runoff would originate from impervious surfaces such as roofs, roads and other paved surfaces. However during relatively high intensity rainfall, the pervious surfaces too could contribute to surface runoff and to pollution loadings. Pollutant wash-off processes are discussed in greater detail in Section 3.2.3.

3.2.3 Pollutant Build-up and Wash-off

The typical approach adopted in stormwater quality modelling is a two stage process replicating pollutant build-up and wash-off. Build-up is the accumulation of pollutants on surfaces resulting from dry and wet deposition during dry periods between rainfall events. Wash-off is the process by which accumulated pollutants are removed from catchment surfaces by rainfall and runoff and incorporated in stormwater flow (Vaze & Chiew 2002). In the context of general understanding and mathematical modelling of these processes, the concept of 'antecedent dry period' in the case of pollutant build-up and 'first flush' in the case of pollutant wash-off are allocated important roles. Therefore these concepts too are discussed in detail below.

A. Pollutant Build-up

Pollutant build-up on a catchment surface is a dynamic process. At any given point in time, there is dynamic equilibrium between pollutant deposition and removal and between pollutant sources and sinks (Duncan 1995). The various pollutant sources have been discussed in details in Section 3.2.1 and pollutant pathways between sources and sinks in Section 3.2.2.

Pollutant build-up and concentration is dependent on the following primary parameters, but the degree of influence they exert is highly variable:

- climate
- land use
- impervious fraction
- population density
- age of the urban area
- landscaping
- average daily traffic
- fraction of area as street surfaces
- pavement material and condition
- days since last rain
- days since streets were cleaned
- method of street cleaning

(Bradford 1977; Pitt 1979; Sartor & Boyd 1972).

Theoretically, the deposition of pollutants should be randomly distributed over street surfaces and other areas. However various removal processes such as wind, vehicle induced turbulence, decomposition, street sweeping and wash-off by rain will constantly impact on the build-up process (Pitt 1979). Figure 3 below provided by Sartor and Boyd (1972) illustrate an idealised view of pollutant build-up on a street surface.

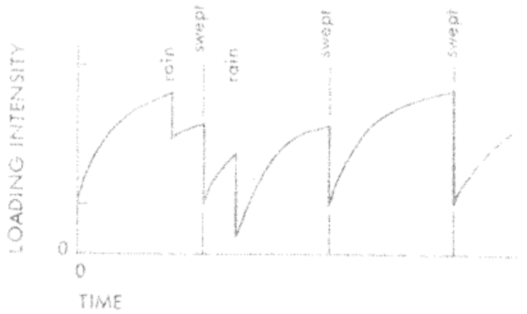

Figure 3 – An idealised view of pollutant build-up on a street surface (adapted from Sartor and Boyd 1972)

Material removed by wind and eddies will either be re-deposited in other areas with more quiescent conditions, re-entrained into the atmosphere or trapped by vegetation (Duncan 1995; Novotny et al. 1985; Pitt 1979). As the system is in dynamic equilibrium, a large departure from equilibrium such as street sweeping or rainfall will generate a larger restoring effort. This explains the curvature of the build-up function as illustrated in Figure 3. Consequently, this means that as soon as a street is cleaned, the faster it will get dirty again by the re-distribution of material from surrounding areas (Novotny et al. 1985).

Despite the descriptive definition of pollutant build-up, the mathematical modelling of this process is not an easy task. It has typically been treated as a linear, exponential, power, log-normal or stochastic function (Baffaut & Delleur 1990; Charbeneau & Barrett 1998; Grottker 1987; Haiping & Yamada 1998; Kuo et al. 1993; Tai 1991; Vaze & Chiew 2002). However limited data sets and the large data scatter makes the form of the relationships hard to determine (Duncan 1995; Whipple et al. 1974).

The data obtained from various research studies clearly confirm the fact that urban stormwater runoff is polluted. However the use of this data for quantitative analysis to describe the processes of pollutant availability on surfaces and its incorporation into stormwater runoff faces two major constraints.

Firstly, it is the difficulty in the mathematical formulation of key anthropogenic activities. Pollution in urban areas vary with anthropogenic related activities such as, concentration of population, commerce and industry and only incidentally with quantitative variables such as land use and average daily traffic (Sartor & Boyd 1972; Novotny & Goodrich-Mahoney 1978; Whipple et al. 1974). Novotny and Goodrich-Mahoney (1978) have recommended that pollutant build-up should not be simply related to land use but to a range of causative factors as noted above. However the degree of influence these factors impart is highly variable and debateable. As an example, Bradford (1977) has noted, that some evidence suggests that average daily traffic correlates with higher pollutant loading rates such as heavy metal concentrations, whilst other evidence suggests the contrary, with solids blown away faster due to increased traffic. Presumably beyond a threshold value, average daily traffic may cease to be an important parameter. In fact, Sartor and Boyd (1972) found that traffic speed, traffic density and parking density to have some influence on street surface contaminants, but no consistent trends could be identified. They have postulated that more dominant factors such as land use and season would have greater impacts.

Secondly, it is the questionable mathematical formulation of key assumptions such as:

1. the pollutant build-up increases with the antecedent dry period (Barbe et al. 1996; Bujon et al. 1992). However this assumption has been questioned by other researchers (Novotny et al. 1985; Whipple et al. 1977). This issue is discussed in detail in Section 3.2.3B.
2. the pollutant build-up starts from zero after a rain event (Irish et al. 1998). This is a convenient assumption for model calibration as pollutant accumulation characteristics can be inferred from measurements of pollutant wash-off. It also implies that pollutant wash-off is source limiting. Figure 4a provided by Vaze and Chiew (2002) illustrates this concept. However other research studies have provided evidence to the contrary as discussed in detail in Section 3.2.3C.

Taking into consideration the above, an alternative concept to pollutant build-up is that storm runoff removes only a fraction of the pollutant load. This implies that pollutant wash-off is transport limiting (Chiew et al. 1997; Hoffman et al. 1984; Malmquist 1978; Vaze and Chiew 2002). This concept is discussed further in Section 3.2.3C under pollutant wash-off. The build-up then occurs relatively quickly to return the surface pollutant load back to the level before the event within a few days. Therefore together with the re-distribution of pollutants, it would result in a catchment surface having a similar amount of pollutant load most of the time. (Novotny et al. 1985; Vaze & Chiew 2002). This assumption has important implications for water quality modelling. It precludes the need for detailed understanding of pollutant build-up. Yuan et al. (2001) refer to a 'loading capacity' for an area where solids deposition and removal are equal after a period of time and the accumulation process will then stop.

Shaheen (1975) has postulated a slightly different scenario for road surface pollutant build-up. Deposition of traffic-related materials will occur at a constant rate under a given set of conditions. At the same time, non traffic-related materials such as litter will be deposited at a linear rate. However, though deposition is uniform, the materials do not accumulate on the roadway surface at a linear rate. Accumulated loads will begin to level off substantially after several days. This has been attributed to passing traffic picking up materials and to 'other processes' which have not been identified by the author.

Incidentally, the studies by Chiew et al. (1997) and Vaze and Chiew (2002) discussed above were undertaken in Melbourne where the rainfall intensities are much less than in South East Queensland. Therefore it is likely that some rainfall events in South East Queensland could be supply limited rather than transport limited in terms of the pollutant load. This further underlies the strong location specific nature of urban water quality and the need to exercise care in the transposition of research outcomes from other geographic regions. Incidentally, Sartor and Boyd (1972) in their extensive study of street surface contaminants in the United States found that loading intensities varied significantly between different cities and land uses.

B. Antecedent dry period

This concept was discussed briefly under pollutant build-up. However due to the fact that there is wide disagreement on the influence of antecedent dry period on pollutant build-up and wash-off, it is considered important to discuss this issue in further detail.

The concept that pollutant build-up is influenced by the antecedent dry period is fundamental to most water quality studies. However it is the nature of the relationship that has proved contentious.

Based on extensive experimental investigations, Sartor and Boyd (1972), Yamada et al. (1993) have confirmed the relationship between pollutant build-up and the antecedent dry period. Further, Sartor and Boyd (1972) have proposed a decreasing rate of increase model for the pollutant build-up curve which is asymptotic to the horizontal as shown in Figure 3. This would imply that at some point in time, the pollutant build-up would be in dynamic equilibrium with removal processes and until a rainfall or street sweeping event would take place.

Yamada et al. (1993) have not specified a specific pollutant build-up curve. However the stochastic data analysis undertaken by them confirmed that the accumulated load tends towards a limiting value after a few days subsequent to a rainfall event. This can be interpreted to mean that the relationship between pollutant build-up and antecedent dry days would be similar to that proposed by Sartor and Boyd (1972). Similarly Charbeneau and Barrett (1998), Grottker (1987), LeBoutillier et al. (2000), Terstriep et al. (1980) for their model studies have adopted exponential curves for pollutant build-up which closely approximates the relationship proposed by Sartor and Boyd (1972).

However the adoption of an exponential relationship is not universal. As an example, Barbe et al. (1996) adopted a linear relationship for pollutant build-up for their modelling studies. Bujon et al. (1992) whilst assuming a linear relationship with the antecedent dry period for pollutant build-up, have also included a decomposition factor with pollutants being removed as a function of the pollutant mass already deposited on the ground.

The observations by Chui (1997) adds a further degree of complexity to the understanding of the pollutant build-up process. Studying two small urban catchments of 107ha and 62ha extent, he found that there was a strong correlation between TSS and COD event mean concentrations and the antecedent dry period. However there was no distinct relationship between TSS and COD loads and the antecedent conditions. The pollutant loads were found to be more closely related to the rainfall characteristics than to the dry weather period.

Along with pollutant build-up, an associated concept commonly adopted is that the pollutant load incorporated in stormwater runoff is determined by the antecedent dry period. Quite often these two concepts are used interchangeably. As examples, Fulcher (1994) and Irish et al. (1998) have assumed a linear relationship between pollutant wash-off and antecedent dry period together with other variables in developing regression equations. However, the validity of this concept has been questioned by numerous other researchers. As examples, Bedient et al. (1980), Ellis et al. (1986), Hoffman et al. (1982, 1984), Weibel et al. (1964) and Whipple et al. (1974) have found that there is no significant correlation between antecedent dry period and stormwater runoff quality.

Bedient et al. (1980) have postulated that these contradictory observations and the resulting modelling approaches could be attributed to the regional character of stormwater response. These concepts are discussed further under Section 3.2.3C, Pollutant Wash-off. The conflicting findings and modelling approaches discussed above only serve to highlight the location specific nature of the processes involved and the significant limitations of a generalised extrapolation of the data.

C. Pollutant wash-off

Wash-off is the process by which pollutants built up on the surface during the preceding dry period is incorporated into the stormwater runoff. As Bujon et al. (1992) have noted, pollutant wash-off incorporates two phenomena which takes place simultaneously. Firstly, as rain falls on the ground it will initially wet the surface and begin to dissolve available water soluble pollutants. The impacting raindrops and horizontal sheet flow provide the necessary turbulence for dissolving the soluble fraction. Secondly, there is the detachment of pollutants under rainfall impact and their transportation by surface runoff. The particulate matter is dislodged by the impact of raindrops and the turbulence created by the horizontal sheet flow will keep them in a form of suspension. As rainfall continues, surface runoff is initiated. The increased flow rate and velocity will begin to move the particulate fraction and will also carry the dissolved pollutants with it to the receiving environment. The particulates will be either suspended in the flow or roll along the ground surface, depending on flow velocity (Overton & Meadow 1996). Though these mechanisms can easily be explained qualitatively, their mathematical formulation and quantitative description is far more complex.

Pollutant wash-off is influenced by factors such as rainfall intensity, rainfall volume and runoff rate (Vaze & Chiew 2002). Chui (1997) showed that event mean concentrations of COD and TSS increases with increasing rainfall intensity. This can be attributed to the fact that storms with a higher rainfall intensity have a greater capacity to scour materials deposited on surfaces and for transport. It can be concluded that for storms with a higher rainfall depth, the total amount of pollutant load washed off will be larger. However Bruwer (1981) observed that relationships established between constituent concentrations and flow rate was of little practical use for predictive purposes. The variations in constituent concentrations could only be explained by the variation in flow rate only to a minor extent.

Also most importantly, pollutant wash-off is influenced by the quantum and the characteristics of the pollutants available, which in turn is affected by the preceding build-up process. Consequently, there is a very strong interaction between pollutant build-up and wash-off (Duncan 1995). However at the same time it is important that the distinction between the two processes is clearly understood. This is not always the case, with some researchers using build-up and wash-off interchangeably in water quality modelling as noted in Section 3.2.3B.

Hoffman et al. (1984), Malmquist (1978), Reinertsen (1981) and Vaze and Chiew (2002) through experimental studies and Chiew et al. (1997) through a modelling study have shown that storm runoff typically removes only a portion of the pollution load. Malmquist (1976) repeatedly flushed an urban street in Goteborg, Sweden using a water tanker which simulated runoff from a high intensity rainfall event. The pollutant concentrations were found to decrease only after the third flush. Based on these research findings, the influence of the antecedent dry period on pollutant wash-off is much less clear than in the case of pollutant build-up. Although some researchers have reported some form of a relationship, the effect is always described as small or qualified in some manner (for example Yamada et al. 1993). However, Hoffman et al. (1982) and Weeks (1981) for example found that there was no significant effect.

Hoffman et al. (1984) found that hydrocarbons in runoff increased with increasing rainfall. This would suggest that an adequate supply of hydrocarbons is generally always available for incorporation into the runoff. Simpson and Stone (1988) noted that the export coefficients for pollutants are mainly functions of the runoff amount and that no single value can be considered typical for a catchment. Consequently a range of values was found to be applicable to cover different rainfall regimes.

Based on field measurements Vaze and Chiew (2002) have proposed two possible alternative wash-off concepts as illustrated in Figure 4. These have been classified as source limiting (Figure 4a) and transport limiting (Figure 4b).

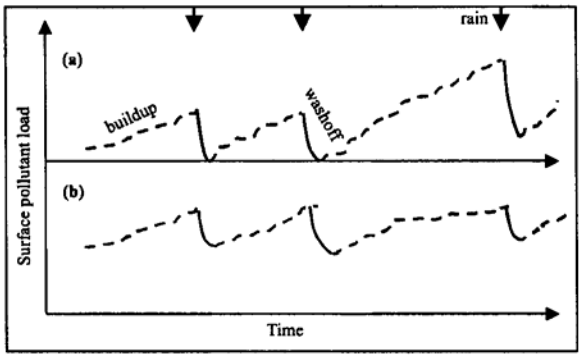

Figure 4 – Hypothetical representations of surface pollutant load over time (adapted from Vaze and Chiew 2002)

The location specific nature of the governing wash-off model is evident from the observations by Driver and Troutman (1989). They developed a general regression equation for long term annual or seasonal pollutant load estimation for urban catchments in the United States. It was found that the most accurate results were obtained for arid regions and the least accurate results for wetter areas. They have attributed this to the fact that in arid regions the model was not supply limited but rather transport limited. Conversely, in the humid regions with higher rainfall volumes, the pollutant accumulation can be washed off completely by more frequent storms and the model would be supply limited. This would result in succeeding storms producing the same runoff rate but relatively smaller pollutant loads.

Management practices such as street sweeping can also appreciably influence pollutant wash-off. Significant investigations into the effectiveness of street sweeping as a pollutant abatement measure were undertaken by Sartor and Boyd (1972) and Pitt

(1979). These studies and also numerous other studies have confirmed that street sweeping is generally ineffective in improving stormwater runoff quality. Street sweeping can remove a relatively large fraction of the coarse particulates built up on a street surface. However it cannot remove the relatively smaller particulates with which most contaminants such as hydrocarbons and heavy metals are associated (Dempsey et al. 1993; Hoffman et al. 1982). The important role played by fine particulates is discussed in detail in Section 4.2. As Pitt (1979) and Vaze and Chiew (2002) have confirmed, street sweeping will release but not remove part of the fixed load consisting of fine particulates, thereby making them readily available for wash-off by the next rainfall event.

Land use and land cover characteristics can significantly influence the presence of pollutant concentrations stormwater runoff. As noted in Sections 3.2.1 and 3.2.2, land use affects pollutant build-up and this is essentially mirrored in the wash-off. This has been dealt with in detail in the above noted Sections. However to briefly mention the salient features:

- sediment wash-off load can be 10 – 100 times greater at construction sites when compared to other land uses (Konno & Nonomura 1981; Line et al. 2002).
- In terms of land use, pollutant load in wash-off increases from commercial to residential and is the highest for industrial areas (Line et al. 2002; Sartor & Boyd 1972).
- In terms of land cover, street surfaces and parking areas are the most critical sources for pollutant generation (Bannerman et al. 1993; Pitt 1979; Sartor & Boyd 1972; Smith et al. 2000).
- In terms of street paving material, asphalt surfaces contribute higher loadings when compared to concrete surfaces (Sartor & Boyd 1972). The asphalt surface itself can be a contributor to wash-off contamination. As Hoffman et al. (1984) have noted, asphalt particles are a significant source of hydrocarbons in runoff.
- In terms of the paved area conditions, poorly maintained areas contribute a relatively higher load when compared to well maintained areas (Sartor & Boyd 1972).

Pollutant wash-off is commonly modelled as an exponential decay function of the available surface pollutant load (for example Baffaut and Delleur 1990; Bujon et al. 1992; Haiping & Yamada 1998; Terstriep et al. 1980). The exponential equation has been assumed to be of various forms and some of the common approaches are listed below:

- function of the runoff rate and the pollutant remaining (Baffaut & Delleur 1990; Charbeneau & Barrett 1998; Haiping & Yamada 1998; Hoffman et al. 1982).
- function of the effective rainfall and the pollutants remaining (Grottker 1987)
- function of the rainfall intensity and the pollutants remaining (Bujon et al. 1992; Terstriep et al. 1980).

However as Duncan (1995) has pointed out, a number of significant problems arise in the use of the exponential form for pollutant wash-off. Firstly, an exponential wash-off function cannot simulate an increase in concentration at any time during a storm. This would be a situation where a higher order storm can lead to enhanced pollutant detachment rather than a proportionate increase.

According to Duncan (1995) and Chiew et al. (1997), the diversity of approaches in mathematically defining wash-off can be attributed to the primary factors which influence this phenomenon. This includes the four explanatory variables; rainfall rate and volume and runoff rate and volume and the main processes; shear stress generated by flow and the energy input by raindrops. The four explanatory variables are correlated to each other and it is difficult to discriminate accurately between them. Also as Duncan (1995) further postulates, it is possible that different processes dominate under different conditions or at different scales.

D. First Flush

As reported by numerous researchers, the ‘first flush’ has been noted as an important and distinctive phenomenon within pollutant wash-off. The first flush produces higher pollutant concentrations early in the runoff event and a concentration peak preceding the peak flow (Deletic 1998; Duncan 1995; Lee et al. 2002). The first flush has also been reported in the case of roof runoff. Quek and Forster (1993) found that the extent of this phenomenon was influenced by roof material, rainfall pH, surface roughness, roof angle and antecedent dry period.

The first flush has significant economic implications in relation to the management and treatment of urban stormwater runoff. The economic significance stems from the fact that structural measures for water quality control facilities such as detention/retention basins are often designed for the initial component of urban runoff. Similarly, rainwater tanks for use as a drinking water resource commonly have a by-pass arrangement for the initial runoff component. Therefore it is important that an in-depth understanding is developed of this occurrence within the overall context of pollutant wash-off.

Hall and Ellis (1985) have claimed that the first flush phenomenon is over emphasised and only 60–80% of storms exhibit an early flushing regime with particularly delayed flushing of metals being common. Other researchers too have observed that the first flush is very frequent in urban runoff, but not necessarily always (Angino et al. 1972; Cordery 1977; Furumai et al. 2002; Helsel et al. 1979; Hunter et al. 1979; Lopes & Fossum 1995; Simpson & Stone 1988). Harrison and Wilson (1985) also concur, noting that the first flush is not a constant feature for all storms in respect of ionic components and it is influenced by the rainfall pattern over the catchment area. Sonzongni et al. (1980) have further strengthened this argument based on their study of urban areas in the Great Lakes region in the US/Canada. They reported that there was no evidence of first flush. However there is no mention of the extent of urbanisation or the catchment sizes that were investigated.

The first flush phenomenon has been investigated for several different contributing components of an urban catchment such as roof runoff, discharge from separate and combined sewer systems and surface runoff. However as Deletic (1998) has pointed out, in view of the diverse definitions, varying sampling strategies and data collection methods, it is difficult to compare results from different studies. This could possibly explain the differences in reported observations in relation to the occurrence of the first flush. Though its occurrence has been confirmed in numerous instances, the observations noted are not consistent. Another confusing issue in relation to the first flush is that numerous researchers have reported widely divergent behaviour of different pollutants.

Hoffman et al. (1984) monitoring four different urban land uses, found that in the case of a runoff event with three flow peaks, suspended solids exhibited proportionate peaks which essentially matched each other. Also the total particulate hydrocarbons mirrored the behaviour of suspended solids. However individual hydrocarbons did not, with one species showing a peak during the initial flush and another species only during the last flush. The authors have attributed this divergent behaviour to causes such as solubility, volatility, susceptibility to degradation and differences in suspended solids particle size distribution which favour adsorption of hydrocarbons and factors influencing supply. Similarly, Sansalone and Buchberger (1997a) found that depending on rainfall intensity, only some particulate bound heavy metals exhibited a first flush. Copper (Cu) was most likely to exhibit a first flush whilst Cd was the least likely to do so.

Hall and Anderson (1986) monitoring a single storm at a commercial land use site found that during the initial stage a large proportion of particulate material was being transported. The soluble material was transported during the middle of the storm event. In the case of the dissolved fraction, Cd exhibited the most pronounced first flush effect followed by Zn and then Cu. They have hypothesised that the timing of transport of these soluble materials during the storm event could be a function of factors such as solubility equilibria, exchange capacity and adsorption-desorption processes associated with solid materials. Lopes and Fossum (1995) have also confirmed the selective first flush behaviour in relation to only some dissolved trace metals. Harrison and Wilson (1985) have noted that the physico-chemical associations in which pollutants are present will also exert a strong influence on the first flush effect.

Hoffman et al. (1985) evaluating highway runoff found that all the monitored pollutants including heavy metals, hydrocarbons and suspended solids responded with high concentrations during the first flush. However most of these pollutants also exhibited a subsequent concentration peak in response to a second flush during the same runoff event. It is also noteworthy that these concentration peaks coincided with the runoff peak rather than preceding it as described in the classic definition of the first flush. This could be possibly due to the efficient stormwater conveyance system in a highway which would have eliminated any lag between concentration and runoff peaks. The observations noted by Cordery (1977) for three urban catchments in Sydney are similar. A first flush was noted for all three catchments. However in the case of one catchment consisting of a stormwater drain, a significant increase in pollutant concentration was noted whenever the stormwater discharge increased rapidly.

Sansalone et al. (1998) noted that a first flush occurred in all the storms monitored by them, but it was found to be weak for the low flow events. It is possible that part of the first flush could be due to the flushing of pollutants deposited in storm sewers and gully pots during the antecedent storm event. As the flow volume reduces, its carrying capacity will diminish with the resulting deposition of particulates at the lower end of the stormwater drainage system. These would be subsequently re-entrained and transported downstream with the next storm runoff (Gupta & Saul 1996). Incidentally, Delectic (1998) using data generated by two similar asphalt covered urban catchments concluded that a strong first flush effect at the end of a drainage system was unlikely to be caused by a flush of pollutants into the system. It was postulated that this could be due to pollutant transformation and transport processes within the drainage system.

However qualitative descriptions commonly found in literature cannot be used as an appropriate basis to plan structural pollutant abatement measures. A mere increase in pollutant concentration at the beginning of a storm cannot be interpreted in a quantitative manner. In the context of stormwater pollution management, it is the pollutant load rather than pollutant concentration that is of significance.

Despite the increase in pollutant concentration, the pollutant load during the initial phase of runoff could be relatively low when compared to the overall load carried by the runoff event. The study by Cordery (1977) on an urban catchment in Sydney has confirmed this hypothesis. He found that the initial high concentrations at low flow resulted in the movement of 20 kg of SS and 3 kg of BOD during a 35 minute time period. However the subsequent higher flow with lower concentrations resulted in the movement of 1,150 kg of SS and 100 kg of BOD during a time interval of 50 min. Similarly Barrett et al. (1998) found in their study of highway runoff, that the overall effect of the first flush was small or negligible. Therefore under these circumstances whether or not the first flush exists and if so, its characteristics are highly debateable issues (Deletic 1998). It could be postulated that the first flush is only a convenient expression to describe a concentration peak.

In addition to rainfall characteristics, catchment characteristics too have been noted to influence the first flush (Lee et al. 2002). Helsel et al. (1979) found that the average incidence of first flush increases with urbanisation. Goettle and Krauth (1980), Lee and

Bang (2000) and Lee et al. (2002) have concluded that this phenomenon is more pronounced in highly urbanised small catchments rather than in large areas. However Bertrand-Krajewski et al. (1998) based on the outcomes of their study have questioned the commonly accepted concept that the first flush occurs more frequently in small catchments. The influence of the antecedent dry period on the first flush is also debateable. Lee et al. (2002) and Saget et al. (1996) have noted that there was no correlation between the first flush and the antecedent dry period, whilst Gupta and Saul (1996) found that the first flush correlated well with peak rainfall intensity and the antecedent dry period.

In understanding the first flush, the major difficulty arises with respect to defining this phenomenon in a quantitative manner. As Bertrand-Krajewski et al. (1998) and Saget et al. (1996) have pointed out, the problem stems from the fact that the 'initial component of runoff' which carries the first flush is never precisely defined. This is despite its commonly reported occurrence in qualitative terms.

Researchers such as Bertrand-Krajewski et al. (1998) and Saget et al. (1996) have used a very strict definition to describe the phenomenon. They have formulated very prescriptive criteria where a first flush is said to have occurred if 80% of the total pollutant mass is transported by the first 30% of the volume discharged during a runoff event. Helsel et al. (1979), Sansalone et al. (1998) and Weeks (1981) have adopted less restrictive criteria, where a first flush is considered to take place if the cumulative pollutant mass vs time curve is above the cumulative runoff volume vs time curve. Figure 5 adapted from Helsel et al. (1979) illustrates this concept.

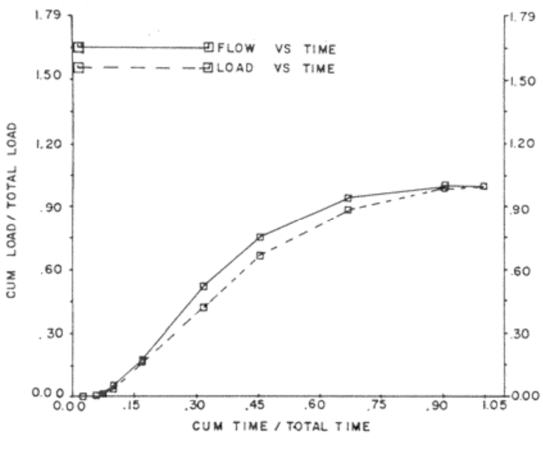

Figure 5 – Variation of incremental load and flow with incremental time (adapted from Helsel et al. 1979)

Ashley et al. (1992) and Gupta and Saul (1996) have used a similar definition to define the first flush in a combined sewer. The first flush has been described as that part of the storm with the maximum divergence between the cumulative percentage of pollutants and the flow plotted against the cumulative percentage of time. This is illustrated in Figure 6.

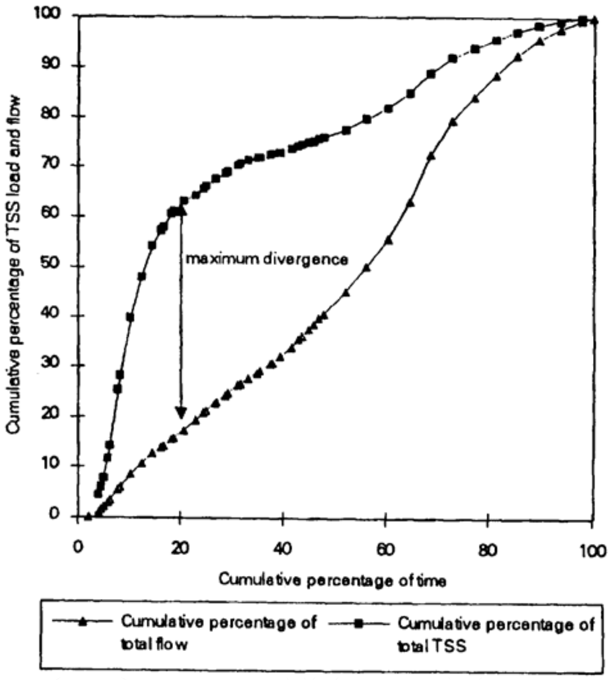

Figure 6 – Definition of first flush based on maximum divergence between cumulative percentage of pollutant and flow (adapted from Gupta and Saul 1996)

The US EPA (1993) has proposed a definition based on a direct comparison with the average dry weather concentration of the pollutant. This is illustrated in Figure 7. The volume V_p corresponding to the first flush is calculated by integrating the runoff curve to the point where the runoff pollutant concentration C(t) equals the average dry weather concentration C_b. This is the shaded area shown in the figure. However as pointed out by Bertrand-Krajewski et al. (1998), this definition has the following significant limitations:

- If the concentration C(t) is higher than C_b for a long period of time, the interception of a large proportion of the total runoff volume would be required. In such circumstances, it is not possible to discuss a first flush volume corresponding to small proportion of the total volume.

- If the maximum value of C(t) is less than C_b, the runoff concentration curve would not be intercepted. This implies that such a discharge is not detrimental to the receiving waters, however large the discharge volume.

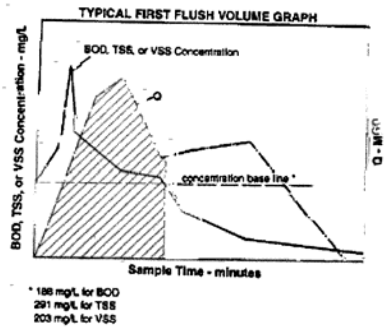

Figure 7 – Definition of first flush based on the average dry weather concentration of the pollutant (adapted from US EPA 1993)

As the above discussion illustrates, the reported results of various studies is confusing and precludes the development of a rational set of concepts to describe the first flush phenomenon. Its occurrence is complex and site specific. As Delectic (1998) has observed, it is clear that the first flush load cannot be calculated using a universal set of rainfall, runoff and climate characteristics or universal types of regression curves.

4. PRIMARY WATER POLLUTANTS

Any review of the environmental impacts of urban stormwater runoff requires consideration of its physical, chemical and biological characteristics. These are directly influenced by anthropogenic activities, catchment and climatic factors and the history of urban development. Consequently, it is difficult to characterise urban runoff. The major pollutant constituents include, litter, sediments, plant nutrients, heavy metals, hydrocarbons, biodegradable organic matter and pathogens. Additionally there can be other contaminants which may be specific to the catchment and land use (Makepeace et al. 1995). The impacts on receiving waters would depend on the type, concentration and the load of these contaminants in the urban runoff. The impacts include:

- aesthetic deterioration
- water quality changes
- public health risk

(House et al. 1993).

The major urban stormwater pollutants other than those of microbiological origin are discussed below in terms of their physico-chemical characteristics, behaviour and impacts.

4.1 Litter

Litter is the most conspicuous category of urban pollution, but is not generally a major source of water pollution. Its foremost impact is visual aesthetics as litter tends to float on the surface. Also another impact of litter is, it can clog the drainage system and thereby impede the flow of stormwater. Unfortunately, due to the high visibility nature of litter, it attracts the most amount of publicity and maintenance effort rather than the more environmentally harmful pollutants.

The primary categories of litter are, packaging materials such as paper, plastic, metal and glass and printed matter such as newspapers and advertising brochures (Sartor & Boyd 1972). These can exist intact or fragmented. Table 3 below gives the street litter accumulation rates provided by Novotny and Goodrich-Mahoney (1978). The origins or the geographical location of this data has not been given. However it does provide an illustration of the quantity of litter that can accumulate on street surfaces alone. Shaheen (1975) has noted that litter averages about 20% of the total weight of materials gathered from roadways. Furthermore it contains substantial amounts of BOD and volatile solids. However due to its large particle size, it does not facilitate easy transport by stormwater runoff. Therefore the impact on receiving waters is generally not significant.

Table 3 – Street litter accumulation rates
(adapted from Novotny and Goodrich-Mahoney (1978)

Land use	Solids accumulation g/kerb m./day
Single family	48
Multiple family	66
Commercial	65
Industrial	127

4.2 Sediments and suspended solids

In the urban environment, sediment is transported from streets and paved areas, rooftops, construction sites and other pervious areas during stormwater runoff. Sediments are transported along flow paths and can be deposited at any time as flow velocities decrease. They would then be available for re-suspension and transport during the next storm event. However some of the finer particulates, generally categorised as suspended solids do not readily settle. They are able to stay in suspension even under quiescent conditions for long periods of time or even forever due to their relatively high surface area/volume ratio and attendant physico-chemical characteristics. Tai (1991) has shown that urban street dust and dirt particles are very stable relative to coagulation processes and do not aggregate into larger, faster settling particles. Sediments can be either solely inorganic or organic in composition or in combination. In this discussion, the term sediments and suspended solids are used interchangeably.

The physical impact of high sediment loadings on biological systems can be very significant. Impacts include, smothering of bottom dwelling fauna and flora and changes to the substrata. High suspended sediment concentrations will reduce water transparency, inhibiting photosynthesis.

However it is the chemical impact on receiving waters that is even more significant. High sediment loads increases the probability of transport of various pollutants by acting as a mobile substrate on which pollutants can absorb, adsorb or otherwise adhere (Harrison & Wilson 1985; Hoffman 1982; Hunter et al. 1979; Sartor & Boyd 1972; Shinya et al. 2000). This primarily relates to the adsorption process which takes place between the liquid and solid interface and is characterised by the attachment of solute molecules to a suspended sorbent solid within an aqueous environment (Tai 1991). The adsorption capacity of a solid particle varies with its structure, size and chemical properties (Pechacek 1994). Sediments therefore have the potential to strongly influence pollutant fate in water environments. Results of the stormwater monitoring undertaken by Lopes et al. (1995) indicated that the total recoverable concentrations of selected heavy metals in urban stormwater were most strongly correlated with the concentration of suspended solids in the samples.

Binding to solid particles is a dominant process among pollutants such as organics and heavy metals. However they differ in their sorption behaviour. Organic compounds including many petroleum hydrocarbons emitted from vehicle exhausts are dominantly adsorbed by organic matter. The presence of other sorbent materials such as clay and metal oxides and properties such as pH are usually less important. However heavy metals show a different selectivity to sorbent materials. The specific binding capacity or the availability of organic matter and metal oxides is important for a number of them. The relevant issues are discussed in detail in Sections 4.4 and 4.5.

High concentrations of pollutants are frequently found in bed sediments. In depositional environments, accumulation of sediments can represent an environmental hazard at the site of deposition or represent a future threat to aquatic environments when they are released into stormwater flow paths by anthropogenic activities, large floods or bioturbation by microbenthos (Bubb & Lester 1994; Field & Pitt 1990; Parker et al. 2000; Warren et al. 2003). This can increase the aqueous concentration and the potential for biological uptake. Polluted material accumulating on the bed of the receiving water is a poor habitat for most aquatic species. Additionally, contaminated in-stream sediments impose severe long term delays on stream recovery rates and the chronic bioaccumulation of toxic materials induces permanent changes to the aquatic community and the ecosystem (House et al. 1993).

In this regard, it is the fine particulates that are of more serious concern. Pollutants such as heavy metals and hydrocarbons are more heavily adsorbed by the fine particulates rather than by coarse particulates (Andral 1999; Bradford 1977; Hoffman et al. 1982; Roger et al. 1998; Sartor & Boyd 1972; Xanthopoulous & Augustin 1992). This is due to the relatively high surface area to volume ratio and the electrostatic charge on the particle surface.

The crucial role played by the fine particulates in urban water pollution is further emphasised by the fact, that it is this fraction that is most easily transported by runoff, will take the longest time to settle and is most easily re-suspended due to any turbulence (Deletic et al. 1997). Furthermore, as Dong et al. (1983, 1984) have pointed out, the longer the particles stay in suspension, the greater the degree of adsorption and desorption of pollutants and the more extensive the biological transformations of degradable components.

The role of clay particles are the most prominent in this regard. Clay particles are complex aluminium silicates that are negatively charged and exhibit a strong affinity towards cations including heavy metals and hydrocarbons. As Warren et al. (2003) have noted, the fine sediments also have a higher organic carbon content than the coarse sediments which further enhances their ability to adsorb hydrophobic pollutants. This is further compounded by the fact that street cleaning equipment is not effective in removing fine particles. As a result fine particles tend to increase in abundance with time on street surfaces (Pitt 1979). Considering the fact that it is the finer fraction that will carry the relatively higher concentration of pollutants, the wash-off of these particulates will only aggravate water pollution. Table 4 presents the results of the study into stormwater particulates by Roger et al. (1998) on a motorway catchment which further emphasis this fact. Similar data has also been reported by Andral (1999), Pechacek (1994) and Vaze and Chiew (2002).

Table 4 – Proportion of sediment sizes by weight (adapted from Roger et al. 1998)

Source	500–1000μm	100–500μm	50–100μm	<50μm
Channel sediment	53.3%	33%	1.6%	12.4%
Runoff water sediment	0.7%	3.9%	9.4%	86%

Nevertheless even though fine particulates adsorb higher concentrations of certain pollutants, the total load should also be taken into consideration. As Marsalek et al. (1997) have noted from the outcomes of their study, the high concentration of metals in the <45μm sediments fraction was not significant as it represented less than 1% of the total mass of solids. These observations have also been confirmed by Pitt (1979) based on a study of street cleaning practices. However the study undertaken by Andral (1999) found that three quarters of the weight of solids in runoff consisted of particles less than 50μm in diameter. Similarly the study by Pechacek (1993) where 89 stormwater samples from residential, commercial and industrial areas were analysed, 90% of the suspended solid particles were found to be less than 10µm with 70% between 1 and 4µm.

Ellis and Revitt (1982) found that 50% by mass of Cd, Cu, Pb and Zn in road sediments is associated with particles less than 500µm. The outcomes from the study by Sartor and Boyd (1972) on street surface contaminants were similar. One of the most important findings of their study was that, a great proportion of the overall pollution potential is associated with the fine solids fraction of the street surface pollutants. Furthermore these fines were found to account for only a minor fraction of the total loading on street surfaces. However as shown in Table 5 below, the very fine silt like material of size <43µm though consisting of only 6% of the total solids contained the major share of most stormwater pollutants. Similar results have also been reported by Ball et al. (1998) and Bradford (1977).

Table 5 – Fraction of pollutants associated with different particle size ranges – percentage by weight (adapted from Sartor & Boyd 1972)

Parameter	<43µm	43–246µm	>246µm
Total solids	6	37.5	56.5
BOD_5	24.3	32.5	43.2
COD	22.7	57.4	19.9
Volatile solids	25.6	34.0	40.4
Phosphates	56.2	36.0	7.8
Nitrates	31.9	45.1	23.0
Kjeldahl nitrogen	18.7	39.8	41.5
Heavy metals	51.2		48.7
Pesticides	73.0		27.0
Polychlorinated biphenyls	34.0		66.0

Based on the results of a monitoring study in a residential area, Xanthopoulos and Augustin (1992) found that heavy metals and PAHs had higher concentrations in the finer fractions of the sediments. They have provided the following classification of stormwater runoff particles as given in Table 6. The first two fractions in Table 6 settle out easily. Additionally, they only play a minor role in pollutant transport. The last two fractions are transported in suspended form and are generally distributed homogenously. They were found to play an important role in the transport of pollutant to the receiving waters.

Table 6 – Classification of stormwater runoff particles (adapted from Xanthopoulos and Augustin 1992)

Fraction size	Classification
> 600μm	Bed load
60–600 μm	Moderately polluted settleable solids
6–60 μm	Highly polluted settleable solids
<6μm	Highly polluted non-settleable solids

The contradictory results discussed above underlie the need to consider the site specific nature of various phenomena associated with stormwater runoff pollution. The data provided by US EPA (1975) on street surface sediments from five US cities confirm the highly variable nature of the particle size distribution. They have attributed these variations to changes in land use, soil and topographic characteristics.

4.3 Plant Nutrients

An important aspect of a water body is aesthetic appeal. Therefore any visual pollution will immediately lower its beneficial value. Nutrients alone do not affect the appearance of water. However, the aquatic growth which they can stimulate can drastically alter not only the visual appearance but also important quality parameters in a water body. Visual impacts may include, colour, turbidity, floating matter and slimes. Other impacts may include, dissolved oxygen depletion and objectionable taste and odour.

The processes taking place in a water body would be virtually impossible to reverse where there is a significant input of nutrients. A closed cycle originates where the nutrients are converted to plant matter and on decomposition is released back into the water environment. The occasional appearance of algae blooms can take place in such water environments (Sartor & Boyd 1972).

The most important and common plant nutrients are nitrogen and phosphorus compounds. Studies have indicated that generally phosphorus is the limiting nutrient or the nutrient which limits vegetation production in freshwater systems. Therefore the absolute and relative concentration of nitrogen and phosphorus can be of significant importance for understanding some aspects of water quality. The TN to TP ratio is highest for forest runoff, less in agricultural and lowest in urban runoff. Therefore the decrease in TN to TP ratio would mean that relatively higher loads of phosphorus is being contributed from urban runoff to receiving waters thereby removing an important constraint to plant growth. The primary source of nutrients include lawn fertiliser, sewer overflows, animal waste, vegetation debris, industrial activities, vehicle exhausts, power generation and atmospheric dry and wet deposition. According to Puckett (1995), during the period 1978 to 1981, in the United States point sources discharged 1.2 million tonnes of nitrogen and 0.26 million tonnes of phosphorus annually. However these values are rendered minor by the estimated 19.4 million tonnes of nitrogen and 5.7 million tonnes of phosphorus discharged from non-point sources. Atmospheric inputs of nitrogen are largest in predominantly urban catchments when compared to agricultural catchments (Puckett 1995).

Lewis (1981) has shown that in a catchment in a tropical climate with marked wet and dry seasons, there is a significant change in the total particulate and nutrient loading. There is an abrupt increase in loading rates at the start of the rainy season followed by a quick decline. However the rates remain higher than those of the dry season. Unfortunately, Lewis (1981) has not attributed any specific reason for these dramatic changes in nutrient concentration in the atmosphere. However this is further confirmation of the claim by Puckett (1995) that atmospheric inputs is an important nitrogen source. In the case of particulate phosphorus, Owens and Walling (2002) have identified the important role of sediments in the delivery system to receiving waters. The deposition of sediments in accumulation zones in floodplains, channels and water bodies result in either temporary or longer-term storage. The subsequent re-mobilisation of the sediment will introduce particulate phosphorus back into the aquatic environment even if contributions from other sources have decreased.

Consequently it is difficult to standardise nutrient export rates for any given land use. This is due to the large number of factors which can influence the interacting processes.

These factors include, soil type, rainfall and runoff characteristics, land use, climatic conditions during the preceding dry period and anthropogenic activities in the catchment. As Artola et al. (1995) have noted, the contribution of nutrients to a waterway is closely related to the specific hydrology of the supplying catchment. To illustrate the complexity and variability of nutrient export data, Line et al. (2002) have quoted data from other studies where the reported annual average export values from urban areas have ranged from 1.6 to 38.5 kg/ha for nitrogen and from 0.03 to 6.23 kg/ha for phosphorus.

Grobler and Silberbauer (1985) have questioned the role of land use in phosphorus export. They found that the only effect of land use was in the relative contribution of point to the non-point sources of TP load. This was based on a study of seven South African catchments over a period of 3-5 years. They found that land use did not play an important part in explaining the variance in phosphorus exports in catchments containing mainly non-point sources. Agreeing with the conclusions by Sonzogni et al. (1980), Grobler and Silberbauer (1985) have identified land form as the controlling factor rather than land use in phosphorus export. Land form has been described as those catchment characteristics which include soil texture, soil type, surficial geology, physiography and soil chemistry. It is not feasible to extend this conclusion universally without wide ranging studies being undertaken. However it could still be argued that urbanisation may provide an indirect impact on phosphorus export resulting from land disturbance and soil erosion due to various anthropogenic activities associated with land use change. Incidentally, the increased export of nutrients from urbanised catchments has been documented in numerous research studies (for example Line et al. 2002; Wahl et al. 1997).

4.4 Heavy Metals

Stormwater runoff from urban areas contain significant loads of metal elements, particularly heavy metals (Davis et al. 2001; Hoffman et al. 1985; Lopes et al. 1995; Pitt 1979; Revitt et al. 1990). As an example, Wilber and Hunter (1979) investigating the impact of urbanisation on a river found that the metal concentrations in the bed sediments increased downstream through the urban area. Heavy metals are of concern because of their potential toxicity. Furthermore unlike most other water pollutants, they do not degrade in the environment.

The important anthropogenic activities which generate heavy metals are, vehicular traffic, combustion of fossil fuels and lubricants and industrial processes. Metallic pollutants generated by industrial processes are industry specific and hence difficult to characterise. In the investigations undertaken by Wilber and Hunter (1979) and Sartor and Boyd (1972), metal concentrations in street sweepings were found to be the highest for the industrial areas. Next to industrial activities, vehicular traffic is an important source for a diverse array of heavy metals. This results from vehicular component wear, fluid leakage and fuel combustion (Revitt et al. 1990; Sartor & Boyd 1972). Table 7 below, adapted from Sansalone and Buchberger (1997b) illustrates the various components and origins of heavy metals from traffic related activities.

Table 7 – Sources of heavy metals from traffic related activities (adapted from Sansalone & Buchberger 1997b)

Component	Brakes	Tyres	Frame & body	Fuel & oils
Cd				
Cr				
Cu				
Fe				
Pb				
Ni				
V				
Zn				

Another appreciable source of heavy metals is corrosion by-products, particularly from roofs (Bannerman et al. 1993). Thomas and Greene (1993) found that the pollutants in roof runoff could be related to the surrounding land use. In industrial areas, there is a relatively high concentration of lead (Pb) and zinc (Zn). They have attributed this to the extensive use of these metallic products in buildings and other uses in industrial areas and their consequent leaching into the environment. Thomas and Greene (1993) also noted the relatively low pH levels of the rainwater in industrial areas which was attributed to the higher concentrations of atmospheric carbon dioxide. Acidic rainfall would enhance the potential for the leaching of metals from building components.

The increase in metal loadings with decreasing pH were confirmed by He et al. (2001) in their study on Cu and Zn in stormwater runoff during first flush and steady-state conditions. Cu sheeting is commonly used in flashing and other trim in roof areas. They also noted that during a rain event, the most easily soluble or poorly adhesive corrosion products are released during the first flush, followed by a steady-state rate. The mobility of heavy metals is dominantly influenced by pH. A lowering of pH results in an increase in the amount of heavy metals in the liquid phase. As Tai (1991) has pointed out, the pH of rainfall has a significant impact on the desorption of pollutants adsorbed on particulates. Furthermore, the ratio of trace metals released at pH 6 against pH 8.1 for similar suspension concentrations, was found to be about 180 for Zn, 45 for Pb and 25 for iron (Fe).

The investigation by Davis et al. (2001) into Pb, Cu, Zn and Cd sources identified that other than for zinc, non-residential buildings produced higher concentrations than residential buildings. This could be attributed to the fact that building construction with metallic components is more common in non-residential buildings. They also found that the sides of buildings were responsible for the generation for an appreciable proportion of these metallic elements. The clear differences between buildings of different materials suggested that the building material itself was the metal source rather than the buildings collecting atmospheric depositions.

However in terms of the detrimental impacts of heavy metals, the toxic effect of a given metal in an aquatic environment is dependent on a number of factors. One of the most important is the form of the particular metal. Most research studies generally report on the total amount of metals present without regard to their physical or chemical state such as whether they are tied up into complex inorganic or organic compounds (Sartor & Boyd 1972). These are the factors which influence the bioavailability of a metal in water.

The partitioning of metal elements between dissolved and particulate-bound fractions varies between different metals. (Hamilton et al. 1984; Marsalek et al. 1997; Morrison et al. 1984; Morrison et al. 1990; Revitt et al. 1990; Sansalone et al. 1996). It is a function of the chemical form in which the metal element occurs and its solubility. It is also influenced by rainfall characteristics such as intensity, pH, nature and quantity of suspended solids and organic carbon present. Sansalone and Buchberger (1997a) have also noted that the average pavement residence time (APRT) has a significant influence on metal dissolution. A high value was found to result in a larger metal element dissolved fraction. They have defined APRT as the time between centroids of the rainfall hyetograph and the runoff hydrograph. This in other words is the same as catchment lag used in surface water hydrology. However a specific reason has not been attributed to this occurrence.

In an investigation into runoff from an urban highway by Sansalone et al. (1996), Cu, Cd, Zn, Ni (Nickel) were found to be mainly in dissolved form whilst Al and Fe were primarily in particulate bound form and Cr and Pb partitioning was intermediate to these two situations. However Hamilton et al. (1984) found that Cu was mainly in the strongly bound organic and residual phases. Dassenakis et al. (1998) in their study of a small Mediterranean river flowing through an urban area found that the concentrations of Cu, Pb, Ni, Cr and Zn were higher in the dissolved phase than in the particulate phase. These contradictory findings serve to emphasise the presence of many factors which influence the proportioning of metal fractions. In the context of heavy metal concentrations in urban runoff, the first flush behaviour of the dissolved and particulate bound fractions as discussed in detail in Section 3.2.3D also needs to be taken into consideration.

Qu and Kelderman (2001) have noted that there is good correlation between most heavy metals such as Cu, Ni, Cd and Zn and organic matter content. These metals interact in solution with dissolved organic matter leading to chelation or complexation processes and are concentrated by adsorption to fine particulates. Charlesworth and Lee (1999a) and Ellis and Revitt (1982) have confirmed metal binding resulting from complexing organic and humic substances. However they also note that street surface runoff possesses high bacterial levels which can lead to subsequent mobilisation of metals from sediments leading to the enrichment of the soluble phase. Similarly those bound to carbonates can be subsequently release through a change in pH (Charlesworth & Lees 1999b). Therefore partitioning of metals elements between dissolved and particulate bound fractions is a dynamic process and concentrations between different fractions may change at various stages along the transport system.

Morrison et al. (1990) provides a concise description of the chemical and physical processes which affect metal speciation. Low pH rainfall which is common in most urban areas assists in the solubilisation of dust associated metals such as in roadways. Most of these dissolved metals will remain as free ionic and weakly complexed inorganic/organic species in surface runoff. During transport over the road surface and the drainage system, ionic metal concentrations are significantly affected if there are changes in parameters such as pH and dissolved organic carbon concentration. Rainwater is rapidly neutralised on contact with salts associated with dust. As a direct result of the rise in pH, ionic forms of Cu and Pb will rapidly adsorb to suspended solids although Cd and Zn tend to remain in solution.

Release of dissolved organic carbon in sewer system liquor will result in further removal of ionic Cu and Pb by complexation. Consequently, the ionic forms of metal species found on paved surfaces are transformed to organically complexed and suspended solid associated species by the time they pass through the sewer outflow (Ellis et al. 1987; Flores-Rodriguez et al. 1994; Morrison et al. 1990). The presence of Fe and Mn (Manganese) oxides can also influence the fractioning of heavy metals into dissolved and particulate phases (Charlesworth & Lees 1999a; Lopes & Fossum 1995; Morrison et al. 1984). These oxides may exist in road sediments as cement between particles or as surface coatings on particles. The work by Morrison et al. (1984) showed that they are excellent scavengers of Pb, Zn and Cd. However these oxides are unstable under anoxic conditions and will release the metals to the soluble phase when such conditions are encountered such as in a sewer system. In regards to the particulate bound fraction, Charlesworth and Lees (1999b) have identified the following categories:

- exchangeable or adsorbed trace metals.
- metals bound to detrital carbonates.
- metals co-precipitated with Fe and Mn oxides as coatings on particles or as cements binding sediment particles together.
- metals associated with organic matter, either incorporated into the tissues of living organisms deposited as detritus or a coating covering grains.

- metals trapped in the crystal lattice of primary and secondary minerals.

The high toxicity of Cd together with its tendency to remain as ionic or dissolved species makes this metal a significant threat to water quality as dissolved metal elements are readily bioavailable and very mobile. Though Pb is predominantly associated with suspended solids, it is generally weakly adsorbed to solid surfaces. Therefore Pb may also be released into the water column when suitable conditions for solubilisation occur such as the presence of reducing conditions (Hamilton et al. 1984; Morrison et al. 1990). However the impact of Pb may not be that significant due to its elimination in gasoline. Numerous recent studies have noted the appreciable reduction in lead concentration in urban stormwater (for example Furumai et al. 2002; Legret & Pagotto 1999).

The important role played by suspended solids and particularly the fine particulates in transporting heavy metals has been commonly noted by researchers (for example Hamilton et al. 1984; Revitt et al. 1990; Sansalone & Buchberger 1997b; Wilber & Hunter 1979). This has also been discussed in detail in Section 4.2. However, Dong et al. (1984) has questioned the widely held concept that fine particles have a higher content of sorbed metals. They found that Cr, Cu, Fe and Ni content in the coarse fractions of urban street dust was greater than in the case of the fine fraction. It has been hypothesised that these metals could have eroded or abraded from metal surfaces. The presence of metal particles was consistent with the large standard deviation obtained for these elements in the coarse fraction. In the same study by Dong et al. (1984), Al, Cd, Pb and Zn concentrations were found to be higher in the clay fraction than in the coarse fraction.

However in reality, the true impact of fine particulates is open to question. There is no doubt that fine particulates such as clay particles have a relatively large surface area to volume ratio and are able to provide a greater number of sites for the adsorption of heavy metals as discussed in detail in Section 4.2. Furthermore, as Wilber and Hunter (1979) has noted in their study, metal concentrations increase with decreasing particle size. Despite this, the greatest total load was found to occur in the fractions larger than 125μm simply because these fractions made up most of the total sediment load. This highlights the need to differentiate between pollutant concentrations and total pollutant load. This not only applies to heavy metals, but also to other pollutants. A higher concentration does not necessarily lead to a higher load particularly, if the specific particulate volume faction is low.

In terms of pollutant abatement, it is the load that is of importance rather than concentration. Similarly Marsalek et al. (1997) evaluating the stormwater runoff quality from a highway bridge found that metal concentrations were higher in the <45μm size fraction than in the whole-sediment samples. However as this fraction represented less than 1% of the total mass of solids, this enrichment in context was insignificant. Sansalone and Tribouillard (1999) have also confirmed this conclusion based on their study of abraded roadway particles. They found that though the concentration of heavy metals such as Zn, Cd, Pb and Cu increased with decreasing particle size, the total metal element mass was predominantly associated with the midrange to coarser particles. This was because the mass fraction of these particles was once again higher when compared to the finer particles. Similarly, Xanthopoulos and Augustin (1992) have observed that the highest concentrations of heavy metals were in the 8 to 60μm range. They have attributed this to the organic content or coating of particles. Nevertheless, this is not necessarily the case all the time. Sutherland (2003) investigating the presence of Pb in road-deposited sediment observed that <63μm was the single most important fraction with 38% of the mass component deposited in these particles.

The observations by Charlesworth and Lees (1999b) may help to explain these contradictory observations. They found that in the source-transport-deposit cascade, the highest proportion of each heavy metal is bound to different fractions in the sediment regardless of the particle size with the dominant binding sites varying in each compartment between organic matter, carbonates or in combination. Carbonates were found to dominate in the binding of Cd, the organic fraction in the case of Ni and Pb and a combination of organic matter and carbonates bind Zn and Cu. This would mean that changes in environmental conditions could lead to release of different heavy metals into the water column. Therefore given the dynamic nature of the urban environment subsequent processes may transfer the metal between different binding sites.

4.5 Hydrocarbons

Urban stormwater runoff plays a significant role in the transport of hydrocarbons to surface water bodies (Hoffman et al. 1982, 1984, 1985: Hunter et al. 1979; Larkin & Hall 1998; Latimer et al. 1990). Hydrocarbons can originate from both anthropogenic and natural sources (Bomboi & Hernandez 1991; Fam et al. 1987; Kucklick et al. 1997; Larkin & Hall 1998; Van Metre et al. 2000). Natural sources generally include the anaerobic degradation of organic materials and forest fires. However, naturally occurring hydrocarbons have been shown to be only a minor contributor to urban runoff under normal circumstances (Stenstrom et al. 1984). It is the anthropogenic sources and the various resulting by-products which are of concern (Fam et al. 1987). As Ngabe et al. (2000) have identified, the concentrations of PAHs in stormwater runoff increases with increasing population densities.

The following discussion will primarily focus on polycyclic aromatic hydrocarbons (PAHs). These products are ubiquitous to the urban environment, and represent the largest class of suspected carcinogens. Additionally, these compounds are lipophilic and may also accumulate in vegetation (Grynkiewicz et al. 2002; Van Metre et al. 2000). Furthermore, as Kucklick et al. (1997) have noted, PAHs unlike many other organic pollutants such as Polychlorinated PCBs, DDTs and Chlorodanes are not declining in urban areas.

The processes inherent in the generation of PAHs are independent of land use. As Latimer et al. (1990) confirmed, the hydrocarbons in urban runoff from industrial, commercial, residential and highway sites have the same general chromatographic characteristics as crankcase oil. Consequently, street runoff which carries street dust can be considered to be the primary source of PAHs to receiving waters. This is in agreement with the findings by Hewitt and Rashed (1991) that PAHs from vehicles are deposited within 20m on either side of a road and the majority is deposited on roadway itself. Street dust would consist of atmospheric dust, road-wear particles and automotive oils and greases (Bomboi & Hernandez 1991).

Larkin and Hall (1998) investigating a highly urbanised catchment and using impervious area and traffic density as indices for urbanisation, were able to correlate the increase in urbanisation to an increase in hydrocarbon concentrations in streambed sediments. Similarly, Sharma et al. 1997 and Van Metre et al. (2002) have identified the crucial role played by vehicles in the generation and distribution of PAHs. The strong correlation between hydrocarbon concentrations in urban runoff and traffic levels have been reported by numerous researchers (for example Bomboi & Hernandez 1991; Grynkiewicz et al. 2002). An investigation into the accumulation of PAHs in urban and rural vegetation by Wagrowski and Hites (1997) found that PAH burdens in urban vegetation samples were on average ten times higher than in rural samples. These results also confirm that atmospheric PAH burdens are higher near the source areas. Therefore, precipitation in urban areas will contribute appreciable PAH loadings from atmospheric sources to stormwater runoff (Grynkiewicz et al. 2002).

In general PAHs can be categorised into two groups; ‘petrogenic’ and ‘pyrogenic’. Petrogenic PAHs originate from unburnt petroleum products. These compounds have a lesser number of benzene rings, generally two. Pyrogenic PAH are formed in combustion processes and have a larger number of rings (Kucklick et al. 1997; Ngabe et al. 2000). Toxicity increases with the increase in the number of benzene rings. The study by Bomboi and Hernandez (1991) found that in urban runoff, the pyrogenic contribution range from 40 to 83% of the total PAH output as opposed to 5 to 25% of the petrogenic contribution.

The anthropogenic sources of PAHs are primarily vehicles, power generation, industrial processes and refuse incineration (Grynkiewicz et al. 2002; Sharma et al. 1997; Wagrowski & Hites 1997). In the case of motor vehicles, the specific sources of PAHs are, tyre wear, crankcase oil, roadway wear and exhaust (Ngabe et al. 2002; Walker et al. 1999). Used crankcase oil in particular has been identified as a major contributor of PAHs to urban runoff. Pruell and Quinn (1988) and Latimer et al. (1990) having analysed used and unused crankcase oil, found that the unused oil had no PAHs whilst the used oil showed substantial concentrations of two to six ring aromatics. Starting with fresh motor oil, Pruell and Quinn (1988) found that the PAH compounds increased rapidly with usage. The accumulation of PAHs in crankcase oil is illustrated in Figure 8 below.

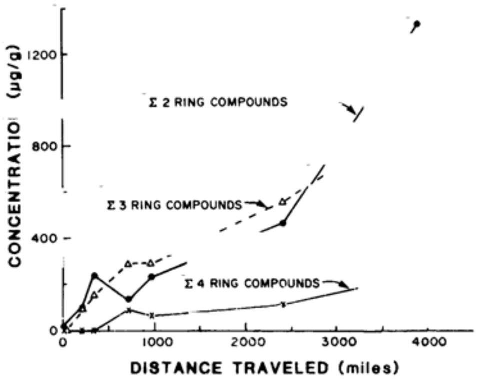

Figure 8 – Concentrations of PAH compounds in crankcase oil (adapted from Pruell and Quinn 1988)

PAHs or for that matter all hydrocarbons are mostly hydrophobic and are easily incorporated into particulates through depositional and partitioning processes. The concentrations and species of PAHs found in sediments have been found to reflect the source characteristics. (Larkin & Hall 1998; Latimer et al. 1990; Makepeace et al. 1995; Moilleron et al. 2002). Kucklick et al. (1997) observed a decreasing gradient of PAH concentrations in the sediments in a water body moving upstream from an urbanised area. Siewicki (1997) and Lake et al. (1979) reported similar findings from their studies. This further reinforces the fact that particulates are the most important transport medium for PAHs. It is further compounded by the fact that urbanisation can increase sediment transport thereby increasing the mobility of hydrocarbons (Larkin & Hall 1998). The distribution of total hydrocarbons associated with particulate material in runoff has been found to range from about 79 to 96% (Gromaire-Mertz 1999; Hoffman et al. 1982, 1984, 1985; Hunter et al. 1979).

The study by Hoffman et al. (1982), though it did not confirm whether there is an enrichment of hydrocarbons on smaller particles, did confirm the importance of the role played by settleable solids in the transport of hydrocarbons particularly during the first flush. The hydrocarbons in the settleable solids were found to represent 64% of the total. Hoffman et al. (1984) based on the analysis of urban runoff from four different land uses found that PAHs were largely associated with two different particle size categories. They have postulated that based on the chemical composition, the larger particle size, 250µm>d>125µm, would have originated from asphalt abrasion. The smaller size, d<45μm, would have originated from atmospheric fallout. Krein and Schorer (2000) derived similar conclusions from their study on road runoff.

An important observation by Latimer et al. (1990) was that the average hydrocarbon concentration on the particulates from all the investigated sources combined was only 20% of the total present in urban runoff particles. This would imply that in addition to the sources investigated, there would be other sources of hydrocarbons in urban runoff which remains to be identified. The sources investigated in the study included, street dust, roadside soil, roadside vegetation and atmospheric deposition. One possibility suggested by Latimer et al. (1990) is the direct dumping of oil by backyard mechanics. They refer to a previous study by one of the co-authors, which found that as much as 19% of the total oil discharged from an urban area into the sewer system could be from this source. Another possible source of crankcase oil to urban runoff, not specifically evaluated in the study was the wash-off from the direct deposition of oil on roads and parking areas.

Analogous increase in PAH concentrations up a sediment core from a lake draining a highly urbanised area was reported by Larkin and Hall (1998). These increases also coincided with the rapid population growth in the study region. Based on these observations it can be postulated that PAHs are rapidly removed from the water column and sequestered into sediments adjacent to sources (Hoffman et al. 1984; Kucklick et al. 1997; Lake et al. 1979; Latimer et al. 1990). Therefore gravity settling can be considered to be easy and effective approach to PAH removal from urban stormwater. However this would also mean that these pollutants could become available once again as discussed in Section 4.2, if the sediments are disturbed due to anthropogenic activities or natural phenomena.

4.6 Organic Carbon

Organic carbon or oxygen demanding materials generally constitute a major pollutant in urban stormwater runoff. The common impact of organic matter is the reduction in dissolved oxygen in water due to microbial oxidation. A minor reduction in dissolved oxygen can usually be tolerated by free flowing water. However if the water body is already stressed due to physical and/or chemical impacts imposed by urbanisation, it could already be low. Therefore a substantial load of oxygen demanding substances could lead to anaerobic conditions resulting in fish kills, foul odours, discolouration and slime growth.

However the more serious impact of organic matter is insidious. Natural colloidal organic matter of size less than <43μm, commonly referred to as dissolved organic carbon (DOC) is responsible for two significant impacts on the distribution of PAHs and heavy metals between aqueous and sediment bound phases. These impacts are termed as 'solubility enhancement' and 'solids concentration effect'. Solubility enhancement is the reduction of the solid-solution partition coefficient which reduces the total sediment sorbed amount, thereby increasing the soluble fraction. The solids concentration effect is where the solid-solution partition coefficient decreases as the ratio of sediment to water is increased. This will result in organic matter in the sediment dissolving into solution and bringing about the solubility enhancement effect described above (Warren et al. 2003).

Additionally, dissolved organic carbon absorbs and reacts with sunlight energy, complexes metals, provides an energy source for microorganisms and associates with hydrophobic substances (Westerhoff & Anning 2000). It also plays a major role in the transport and bioavailability of metals and hydrocarbons through complexation reactions. Furthermore, organic carbon adsorbed on suspended solids particles increases their sorption capacity for combining with hydrophobic organic chemicals and some heavy metals such as Pb and Zn (Parks & Baker 1997; Roger et al. 1998). Though these characteristics may be considered beneficial aspects, the organic matter is liable to microbial decomposition, thereby returning the pollutants back to the dissolved phase. This issue has also been discussed in Section 4.3.

Gromaire-Mertz et al. (1999) and Sartor and Boyd (1972) have identified street surfaces as a major contributor of oxygen demanding materials to receiving waters. The loadings are found to vary over a wide range depending on the characteristics of the urban area and time since the last rainfall or street sweeping. Sartor and Boyd (1972) have also noted that organic materials tend to accumulate on street surfaces much faster than inorganic materials. This would mean that, leaves, litter etc. are dominant over sand and dust-type materials.

Roger et al. (1998) in apportioning sediment from motorway runoff into a range of particle sizes reported that the organic matter concentration was significantly higher for the <50μm fraction when compared to other particle sizes. Using volatile solids as a surrogate for organic matter, Sartor and Boyd (1972) too, found that the finer particulates contained more organic matter than the coarser particles. They have hypothesised that this could be due to the fact that organic matter has low structural strength and can easily be ground into fine particulates. Secondly, non-particulate organic matter will adhere to the surfaces of particles. Therefore as finer particles have a larger unit surface area, relatively more organic mater will adhere. This in turn has implications with respect to the transport of heavy metals and hydrocarbons to receiving waters. Also as Sartor and Boyd (1972) have pointed out, the size range at which sweepers are essentially ineffective have been found to contain between 34-99.5% of the oxygen demand loading. Hence the majority of the organic carbon present on a street will be transported with surface runoff.

5. CORRELATION OF LAND USE WITH POLLUTANT LOADINGS

Many factors affect the quality of stormwater runoff. Frequently cited factors include land use, traffic intensity, rainfall characteristics and climate (Brown 1988; Hall & Anderson 1986; Helsel et al. 1979; Lewis 1981; Sartor & Boyd 1972; Yamada et al. 1993). In terms of anthropogenic related activities, numerous research studies have strived to relate land use to pollutant loadings. The outcomes reported are sometimes conflicting particularly in relation to individual pollutant categories.

Brezonik and Stadelman (2002) evaluating stormwater data from 15 studies covering 68 catchments found that the multiple linear regression models developed by them to predict event mean concentrations had very limited reliability. It is possible that variations in sample collection and testing between different studies and differences in catchment characteristics may have also contributed to some of the uncertainty. Lopes et al. (1995) and Parker et al. (2000) observed that there was no statistically significant relationship between land use and constituent concentrations for an urban region in Arizona, USA.

In the case of individual pollutant species too, research has not been able to show statistically significant relationships but has generally been able point to qualitative relationships with different land uses. The following discussion summarises a number of these research studies in order to illustrate the general trend in observations in this regard.

Owens and Walling (2002) have noted that the range and absolute values of total phosphorus content tend to increase with the increase in the level of urbanisation and industrialisation. Sartor and Boyd (1972) in their comprehensive study of street surface pollutants in a number of cities in the United States, found that in respect of oxygen demanding materials, light industrial sites were the most heavily polluted. Commercial areas were the least polluted followed by residential and then heavy industry. They have postulated that this could be due to spillage of loads from trucking operations as light industrial areas tend to be dominated by warehousing and bulk storage operations. Secondly, street sweeping tends to be biased towards commercial areas at the expense of other land uses. However in the case of total solids, Sartor and Boyd (1972) found that heavy industrial areas to have the highest loading, followed by light industrial, residential and commercial areas to be the least polluted. Table 8 below provides a summary of the findings by Sartor and Boyd (1972) on pollutant loadings on street surfaces for different land uses.

Table 8 – Pollutant load (kg/kerb km)

(adapted from Sartor & Boyd 1972)

Total solids	Industrial	Residential	Commercial
Total solids	795	341	102
BOD_5	6	3	0.9
Total nitrogen	1.16	0.59	0.16
Total phosphorus	0.97	0.31	0.09
Heavy metals	0.22	0.16	0.05

However in respect of individual heavy metals, the data produced by Sartor and Boyd (1972) shows that the distribution appears to be random and the trend in respect of land use disappears. This data is presented in Table 9.

Table 9 – Distribution of heavy metals (% by weight) (adapted from Sartor & Boyd 1972)

Metal	Industrial	Residential	Commercial
Cr	8	5	5
Cu	14	10	20
Zn	44	38	24
Ni	5	1	3
Hg	4	10	20
Pb	25	36	28

In contrast, Liebens (2001) in a study of sediments from stormwater retention ponds found that heavy metal concentrations in commercial areas were consistently higher than in residential areas.

In the case of PAHs, Walker et al. (1999), noted that the distribution of different species varied with the land use. As an example, PAH loading from residential areas tend to be enriched in fluoranthene and low in naphthalene. In the case of suspended solids, Pechacek (1993) noted the general trend in industrial sites contributing higher concentrations to urban runoff than residential and commercial areas. However the same study could not detect a significant relationship between particle size distribution and land use. Similar observations with respect to industrial areas were noted by Latimer et al. (1990) for total hydrocarbons and Hoffman et al. (1984) with respect to PAHs. Hall and Anderson (1988) in their study found that residential areas had the highest chemical oxygen demand loading, with the industrial and commercial sites having similar values. They have postulated that this could be due to the greater amount of vegetal material from gardens, lawns and trees in residential areas when compared to industrial and commercial areas.

Bannerman et al. (1993) and Larkin and Hall (1998) have noted that streets and by association, parking lots are probably the critical source areas for pollutants for most urban land uses. As the results from the study by Sartor and Boyd (1972) on street surface contaminants have shown, it is the difference in pollutant concentrations on street surfaces which directly influence urban stormwater runoff quality from different urban land uses. These observations could also be extended to imply that the extent of street surfaces within a specific urban area would be the single most influential factor dictating urban stormwater runoff quality rather than land use.

Though microbial quality did not form part of the review, the observations by Bannerman et al. (1993) further highlight the complexities inherent to urban stormwater quality. They noted that faecal coliform bacteria count to be higher in runoff from residential areas when compared to commercial and industrial areas. They have attributed this to the presence of wildlife and pets in residential areas and their absence in other land use areas. However Payne and Hedges (1990) concluded that catchment characteristics influence the biological impact on receiving waters. This includes catchment area and land use. Higher impacts were likely with the increase in catchment area size and also for industrial and highway outfall sites.

6. CURRENT STATE OF KNOWLEDGE

6.1 Overview

The following discussion summaries the important conclusions derived from the review in order to provide a sharper focus on the current state of knowledge with respect to urbanisation impacts on the water environment. It brings together the often conflicting research outcomes discussed above so as to clearly demarcate areas where there is a depth of existing knowledge and areas where there is a discernable dearth of research.

6.1.1 Impacts of Urbanisation

The quantity impacts of urbanisation and changes to the hydrologic regime of catchments is well documented in research literature. This can be ascribed to the fact that in the past, the primary interest of regulatory authorities was flood mitigation. The current focus on urban water quality is of relatively recent origin. The most significant quantity impacts are:

- the increase in surface runoff peak and volume and reduced time to peak during rainfall events;
- the altered flow conditions in waterways during dry periods.

The important factors which influence quantity impacts include, the type and extent of impervious cover, the layout of the streamflow network and the spatial distribution of the urban areas. It has also been found that the quantity impacts are significant for more frequent rainfall events and the significance reduces for less frequent events.

However in the case of quality impacts, the research outcomes are often conflicting. Consequently, it is difficult to develop unambiguous cause-effect relationships for urban water quality impacts. There is no question that the urban environment is affected by a variety of anthropogenic activities which introduces numerous pollutants to the environment. However major uncertainties arise in efforts to articulate the process kinetics of pollutant generation, transmission and dispersion. This is further compounded by the fact that the synergies between various causative processes and climatic characteristics are little understood.

The sources and pathways of urban water pollutants have been clearly identified in research literature. However the current state of knowledge with regards to the processes of pollutant build-up and wash-off is extremely limited. The limited data sets available and the large data scatter makes the form of the relationships difficult to determine. These processes are influenced by a range of factors which do not lend themselves to simple mathematical modeling and the simplistic modeling approaches commonly adopted can lead to gross error. Typically these processes are treated as linear, exponential, power, log-normal or stochastic functions.

In the case of pollutant build-up, the difficulty is the mathematical formulation of key anthropogenic activities which influence pollution generation and the various transport and removal mechanisms. Research has shown that pollution in urban areas vary with anthropogenic related activities such as concentration of population, commerce and industry. Additionally, removal processes such as wind, vehicle induced turbulence, decomposition and street sweeping will constantly impact on the build-up process. Furthermore, the material removed by wind and eddies could be re-deposited in other areas.

The major difficulties in modeling pollutant build-up is further compounded by the uncertainty in relation to the role of the antecedent dry period. Based on experimental investigations, various relationships with pollutant build-up have been proposed ranging from a decreasing rate of increase model to a linear model. The limited applicability of these models confirms the strongly location specific nature of the processes involved.

Pollutant wash-off too is no different in terms of the lack of knowledge of the inherent processes. Research has shown that pollutant wash-off is influenced by rainfall characteristics and the preceding build-up process. It has also been found that wash-off characteristics vary for different pollutants. The role of the antecedent dry period is even more uncrear in the case of pollutant wash-off. Although some researchers have reported some form of a relationship, the effect is always described as small or qualified in some manner. Other researchers have found that there is no significant effect.

Pollutant wash-off is commonly modeled as an exponential decay function of the available surface pollutant load and various rainfall or runoff parameters. However the limited validity of this modeling approach has been commonly noted. It has been postulated that this could be due to different processes dominating under different climatic conditions or at different scales. Two alternative concepts have been proposed for pollutant wash-off behaviour. Wash-off is either ‘source limiting’ where all the pollutants are removed due to a rainfall event or ‘transport limiting’ where surface runoff will remove only a portion of pollutants built up on the surface. Research has shown that the validity of these concepts are climatic dependent. In the case of arid regions, a transport limiting model would be applicable whilst in high rainfall areas it would be a source limiting model that would govern.

An important concept within pollutant wash-off is the ‘first flush’. It has been noted as an important and distinctive phenomenon where a high pollutant concentration peak is produced early in the runoff event. The first flush has significant economic implications in relation to the management of urban stormwater. This stems from the fact that structural measures for water quality control are often designed for the initial component of runoff, in other words the first flush.

Despite the claims by numerous researchers of its occurrence during a stormwater runoff event, there has been equally strong claims that it is over emphasized. It has also been claimed that the first flush is not necessarily a common feature, it is influenced by rainfall characteristics or it is only significant in small highly urbanized catchments. A contributing factor to this confusion could be the diverse definitions, varying sampling strategies and data collection methods which make it difficult to compare results from different studies. This is further compounded by the divergent behaviour of different pollutants and between soluble and particulate materials at the commencement of stormwater runoff. It has been hypothesised that the timing of transport of various pollutants could be a function of factors such as solubility equilibria, exchange capacity and adsorption-desorption processes.

An important issue that has been raised in relation to the first flush and in the context of water quality management is that it is the pollutant load rather than pollutant concentration which is of significance. Research has shown that despite the mere increase in pollutant concentration during the initial phase of runoff, the associated pollutant load could be significantly low due to the associated low flow when compared to the overall load carried runoff event. Under these circumstances it is open to question whether the concept of first flush has any technical merit from a water quality perspective and whether it is merely a convenient expression to describe a qualitative phenomenon.

The main problem stems from the fact that the ‘initial component of runoff’ which supposedly carries the first flush is never precisely defined. It is in this context that there have been attempts to define this phenomenon in a quantitative manner. The definition proposed in research literature have included very prescriptive criteria such as, a first flush is said to have occurred if 80% of the total pollutant mass is transported in the first 30% of the volume discharged during a runoff event. Additionally, less restrictive criteria have been proposed where a first flush is considered to have taken place if the time based cumulative pollutant mass curve is above the cumulative runoff volume curve. The major drawback in adopting criteria of this nature is that the occurrence of this phenomenon may have to be assessed on an individual event basis and on a catchment basis.

It can be concluded that the outcomes of various studies is confusing in relation to the first flush. The current state of knowledge precludes the development of a rational set of concepts to describe this phenomenon. Due to its complexity and site specific nature, the first flush load cannot be calculated using a universal set of rules.

6.1.2 Primary Pollutants

The primary water pollutants identified in literature includes, litter, suspended solids, nutrients, heavy metals, hydrocarbons and organic carbon. Litter is the most conspicuous among urban pollutants, but in reality it is not a major source of pollution. Its foremost impact is visual aesthetics and due to this, it tends to attract the most amount of publicity and maintenance effort rather than the more harmful pollutants.

Sediments or suspended solids are the most ubiquitous of urban pollutants. Rather than their physical impact, it is their chemical impact which is of more serious concern. Suspended solid particles act as mobile substrate on which other pollutants such as heavy metals and hydrocarbons will adsorb. Sediments therefore have the potential to strongly influence pollutant fate in water environments. High concentrations of sediments are frequently found in bed sediments. In depositional locations, the accumulation of sediments can represent an environmental hazard if they are released into stormwater flow paths due to any disturbance.

In regards to suspended solids, it is the fine particles which are of more serious concern. It is this fraction that is most easily transported by runoff, will take the longest time to settle and is most easily re-suspended due to any disturbance. Secondly, they have a relatively larger surface area and electrostatic charge on the surface. Furthermore, street cleaning equipment is not effective in their removal. However despite these detrimental characteristics, the overall importance of fine particulates is open to conjecture. Even though fine particulates may adsorb higher concentrations of pollutants, the total load also has to be taken into consideration. A number of research studies have shown that the fine particulates constitute only a small fraction of the total mass of solids thereby negating their overall impact. However other studies have reported completely opposite results with a relatively high proportion of fine particulates or a major share of the pollutant load contained with the finer fraction. These contradictory results emphasises the need to consider the site specific nature of various phenomena associated with stormwater runoff pollution. Secondly it is important to differentiate between pollutant concentration and pollutant load.

Nutrients are an important urban water pollutant. Nutrients alone do not affect water quality, but the aquatic growth they stimulate not only impact on the visual appearance, but also affect other important quality parameters. Due to nutrient impacts, a closed cycle originates, and the processes taking place are virtually impossible to reverse.

The common nutrients are nitrogen and phosphorus. Studies have indicated that phosphorus is the limiting nutrient for vegetation production in freshwater systems. Research has found that the ratio of nitrogen to phosphorus reduces as a catchment land use moves from forest to agriculture to urban. Therefore urbanisation removes an important constraint to vegetation growth in waterways.

However, despite the fact that research literature has widely identified the common sources of nutrients as urban activities, there are numerous other contradictory findings. As an example, atmospheric inputs have been identified as an important nitrogen source. It could be argued that urban activities such as combustion of fossil fuels are a significant contributor to atmospheric nitrogen. However due to various atmospheric processes, the subsequent deposition of these pollutants could be far removed from their point of origin. Also another important point to note is that the primary source of nitrogen input to waterways is atmospheric rather than through pollutant build-up on surfaces. It has also been shown that in tropical climates with marked wet and dry seasons, there is an abrupt increase in nutrient loading rates at the start of the rainy season followed by a quick decline. Hence nitrogen inputs into waterways may not have a direct relationship with a standard urbanisation parameter such as impervious area. Rather it could be more closely related to atmospheric characteristics which are quite complex to model mathematically.

The same uncertainties also exist in the case of phosphorus. Studies have shown that land form as the controlling factor rather than land use in phosphorus export from a catchment. Land use has been described as the catchment characteristic which includes soil texture, soil type, surficial geology, physiography and soil chemistry. However it could be argued that urbanisation provides an indirect impact on phosphorus exports due to land disturbance and soil erosion resulting from various anthropogenic activities associated with land use change. However the mathematical modelling of these processes is not an easy task.

Consequently it is difficult to standardise nutrient export rates for a given land use. This is due to the complexity of factors which influence the interacting processes. The large variations in data values reported in various studies bears testimony to this conclusion.

In terms of urban water pollutants, heavy metals are a major concern due to their persistence in the environment and their toxicity. The major sources of heavy metals are traffic related activities and industrial processes. Corrosion by-products has also been identified as an appreciable source of heavy metals. The important role of acidic rainfall in contributing heavy metals to stormwater runoff has been identified in literature. It not only increases the leaching of metals from building components, but also affects the amount of metals in the liquid phase. This is due to the desorption of metals adsorbed to particulates.

The toxic effect of a given metal in an aquatic environment is dependent on a number of factors, one of which is the form of the particular metal. Most research studies generally report on the amount of metals present irrespective of their physical or chemical state and as to whether they are tied up in complex compounds. These are the factors which dictate the bioavailability of a heavy metal in water.

The partitioning of metal elements between dissolved and particulate bound fractions vary between different species. It is a function of the chemical form in which the metal occurs and is also influenced by runoff characteristics such as pH, nature of suspended solids and organic carbon present. Research studies have commonly reported quite contradictory results for the dissolved and particulate fractions for a specific metal thus emphasising the many factors which influence this proportioning. It is important to note that along the flow path, if there are changes in parameters such as pH, these proportions would change accordingly. As an example, the release of dissolved organic carbon in sewer system liquor will result in the conversion of dissolved forms of some heavy metals to particulate forms which are organically complexed and associated with suspended solids.

The presence of iron and manganese oxides too will contribute to the partitioning of heavy metals into the particulate bound fraction. These oxides are commonly present in road sediments as cement between particles or as surface coatings on particles. However these organically complexed or oxide bound fractions are unstable and can be easily released back into the soluble phase due to aerobic, microbial activity or anoxic conditions in the sewer system. Therefore the partitioning of metal elements between dissolved and particulate bound fractions is a dynamic process and the concentrations between different fractions may change at various stages along the transport system.

The important role played by suspended solids in the transport of heavy metals has been clearly confirmed in research literature. However there is uncertainty with regards to the specific particle size fraction that plays the most prominent role. A research study on the source-transport-deposit cascade may contribute to clarifying some of the contradictory outcomes reported in research literature. The study found that the highest proportion of each heavy metal was bound to different fractions in the sediment regardless of the particle size. The dominant binding sites varied in each compartment between organic matter, carbonates or in combination, and these compounds would dominate different particle sizes. Carbonates were found to dominate the binding of cadmium, the organic fraction in the case of nickel and lead and a combination of organic matter and carbonates in the binding of zinc and copper. However there is no mention about iron or manganese oxides in the study, even though they have been identified in other studies as playing a significant role in the binding of heavy metals to particulates.

Similar to heavy metals, hydrocarbons too are pollutants of significant concern due to their toxicity. Hydrocarbons can originate from either natural or anthropogenic sources. Natural sources include, degradation of organic matter and forest fires. Under normal circumstances naturally occurring hydrocarbons are only a minor contributor to urban runoff. It is the products of anthropogenic activities which are of major concern.

Additionally in an urban environment, it is polycyclic aromatic hydrocarbons (PAHs) which are of major concern. The processes inherent in the generation of PAHs are independent of land use. Research has shown that PAHs in stormwater runoff from different land uses have similar chromatographic characteristics as used crankcase oil. Furthermore, it has been shown that unused crankcase oil has no PAHs, whilst the various PAH compounds increased rapidly with the usage of the oil.

The primary source of PAHs is the combustion of fossil fuels. The crucial role played by vehicles in the generation and distribution of PAHs and the strong correlation between their concentration and traffic levels have been confirmed in research literature. Secondly, it is not only build-up on the catchment surfaces but also atmospheric sources too can contribute to PAH loadings in stormwater runoff.

PAHs are mostly hydrophobic and easily incorporated into particulates through various processes. In a water body, PAHs in the sediments have been found to show a decreasing gradient moving upstream from an urbanised area. This reinforces the fact that sediments are the most important transport medium for PAHs. Also it is compounded by the fact that urbanisation can increase sediment transport, thereby enhancing the mobility of PAHs.

Research studies have noted the important role played by settleable solids in the transport of hydrocarbons. It can be postulated that PAHs are rapidly removed from the water column and sequestered into sediments adjacent to sources. Therefore gravity settling can be considered to be an effective approach to PAH removal from urban stormwater.

Another major pollutant of concern in urban stormwater is organic carbon or oxygen demanding materials. Its most easily recognisable impact is the reduction of dissolved oxygen in water due to microbial oxidation. Consequently if the water body is already stressed due to various impact of urbanisation, its dissolved oxygen level could already be low. Therefore a significant load of oxygen demanding substances could lead to anaerobic conditions resulting in fish kills, foul odours, discolouration and slime growth. However the more serious impact of organic matter is insidious due to the actions of colloidal organic matter of size less than <43μm, which is commonly referred to as dissolved organic carbon (DOC).

In the case of PAHs and heavy metals, DOC leads to the reduction of the solid-solution partition coefficient. This result in the reduction of the amount of pollutants sorbed to sediments, thereby increasing the soluble fraction. DOC absorbs and reacts with sunlight energy, complexes metals, provides an energy source for microorganisms and associates with hydrophobic substances. Additionally, organic carbon adsorbed on suspended solid particles enhances their sorption capacity for combining with hydrocarbons and some heavy metals. Though some of these characteristics can be considered to be beneficial, the organic matter is liable to microbial decomposition, thereby returning the pollutants back into the dissolved phase.

Street surfaces have been identified as a major contributor of oxygen demanding materials, with loadings varying depending on the characteristics of the urban area. Also organic materials have been found to accumulate on street surfaces much faster than inorganic materials. Furthermore, it is the finer sediment particulates which contain more organic matter than coarser particulates. This has significant implications in relation to the transport of heavy metals and hydrocarbons to receiving waters. A more serious impact relates to street cleaning equipment. The size range at which sweepers are relatively ineffective has been found to contain between 34–99.5% of the oxygen demand loading. Therefore the majority of organic carbon present on a street will be transported with surface runoff.

6.1.3 Correlation of Land Use with Pollutant Loadings

A common objective of most urban water quality studies has been to strive to relate land use to pollutant loadings. However the outcomes to-date have been far from conclusive. The major failure has been the inability to derive statistically significant relationships even though qualitative relationships are generally evident.

In terms of individual pollutants, most studies have been able to correlate loadings with land use, albeit qualitatively. However there have been some contradictory results. As an example, heavy metal loadings are lowest for commercial areas, followed by residential areas and heaviest for industrial areas. In fact, this is the general trend described in literature for most pollutants other than for oxygen demanding materials where residential areas have a higher loading due to vegetal materials from gardens. However for individual heavy metal species, this trend disappears and the distribution is random. This is obviously due to various sources and processes which are little known or understood. This type of randomness has also been reported in the case of PAHs. The relatively low pollutant load in commercial areas has been attributed to the greater frequency of street cleaning when compared to other land use areas.

The overriding conclusion in urban water quality studies is that street surfaces and by implication parking lots are the critical source areas for pollutants for most urban land uses. Research has shown that it is the difference in pollutant concentrations on street surfaces which directly influence urban stormwater quality from different land uses. This would imply that the exten of street surfaces within a specific urban area would be the single most influential factor dictating urban stormwater runoff quality. It could provide a primary reason for the inability of various research studies to detect statistically acceptable relationships between urban water quality and land use.

6.2 Management Implications

The management of quantity impacts on stormwater runoff is relatively straight forward. The common approach is the provision of detention/retention basins. These act to retain part of the runoff volume and/or attenuate the runoff hydrograph, with the objective being to replicate the pre-urbanisation hydrograph. Under suitable conditions, these structures have proven to be effective. However it is important to bear in mind that detention/retention basins are feasible only for relatively low average recurrence interval rainfall/runoff events. The provision of detention facilities for higher order events may not be economically feasible. However as research has shown that quantity impacts due to urbanisation are not significant for higher order events, it could be argued that this may not be a serious impediment to the management of urban stormwater quantity impacts.

Unfortunately the management of water quality impacts are far more complex. The provision of appropriate facilities would depend on the targeted pollutants. The removal of pollutants such as litter is relatively simple. However the removal of other pollutants poses a more challenging task. The provision of gross pollutant traps (GPTs) is a common practice in most urban areas. In addition to litter removal, they may also incorporate sediment removal facilities. They may include a screen for litter removal and a sediment trap at the base for sediment removal. However the feasibility of the use of GPTs is open to question due to two significant factors. Firstly, if there is any appreciable time delay in the removal of the collected pollutants, anaerobic conditions could occur in the water collected in the sediment trap due to the decomposition of any organic matter present. Therefore in addition to adverse impacts such as odour, the facility could become a pollutant exporter. Therefore a simple screen or trash rack which will only remove gross pollutants could prove to be more desirable. However the contribution to water quality improvement achieved by the removal of gross pollutants is open to question other than for aesthetic reasons. As Allison et al. (1998) have pointed out, the nutrient contribution by leaf litter is not significant compared to the total nutrient load in stormwater.

The second factor relates to the size range of sediments removed by a sediment trap or for that matter any other treatment measure. Chapter 5 has discussed in detail the significant role played by suspended solids in acting as a mobile substrate for pollutants such as heavy metals and PAHs. As such there is no doubt as to the importance of the removal of suspended solids from urban stormwater runoff. In fact one of the most effective measures for the removal of heavy metals and PAHs from stormwater runoff would be suspended solids separation (Dong et al. 1984; Hunter et al. 1979; Sansalone & Tribouillard 1999). However at the same time it is important that the facilities provided are capable of removing the critical size range of sediments which would be carrying a significant pollutant load. Research has shown that due to their physico-chemical characteristics, the finer particulates are more efficient in the adsorption of pollutants and hence will carry a relatively higher pollutant concentration. Greb and Bannerman (1997) have raised similar concerns in the reduced efficiency of a wet detention pond removing fine particulates. They found that pollutants are removed at a rate less than the suspended solids removal rate, and the removal rates are influenced by influent particle size distribution.

Nevertheless as Hoffman (1982, 1984) have observed, an appreciable component of PAHs in stormwater runoff originate from abraded asphalt particles which are relatively large. Additionally, outcomes from a number of studies have noted that as the fraction of fine particulates in runoff is small, the total pollutant load would be smaller when compared to the load carried by the coarser particulates. Therefore it has been argued that it is the load rather than the concentration which is of importance and hence the focus should be on the removal of the coarser fraction. Contrary to these findings, other studies have reported a larger fraction of fine particulates. These contradictory findings clearly point to the fact it is the catchment characteristics which play the most significant role in urban stormwater runoff quality. Therefore any treatment measures adopted should be designed taking these characteristics into consideration.

The same conclusion applies in relation to pollution source areas. There is overwhelming evidence that street surfaces are among the most important source areas for urban stormwater pollution. Hence the adequate treatment of stormwater runoff from roads can help to alleviate stormwater runoff pollution. As such an important management measure commonly adopted is street sweeping or cleaning. Street cleaning does fulfil an important aesthetic role, but its effectiveness as a water quality improvement measure is much in dispute.

An extensive study by Pitt (1979) into street cleaning found that the frequency, number of passes and street surface condition influence performance more than differences in equipment used. Streets in good condition were found to be easier to keep clean than those in poor condition. The study concluded that street cleaning programs should be varied for different service area conditions. Extensive research undertaken over the years has consistently proved that street sweepers are not particularly capable of removing fine particulates (Pitt 1979; Sartor & Boyd 1972; Vaze & Chiew 2002).

As Vaze and Chiew (2002) have concluded, another issue of concern with street cleaning is that the street sweeper could be releasing the finer materials built up on the surface, but will remove only some of the material. Therefore the remainder of the fine material will be available for wash-off by the next storm. However conclusions by Malmquist (1978) on the effectiveness of street sweeping runs counter to the above observations. Two identical streets, one of which was swept and the other unswept was analysed. The wash water from the unswept street was found to contain 2.3 times more suspended solids and heavy metals than the swept street. Hence sweeping was concluded to have a noticeable effect on pollutants.

Treatment of the first flush is another popular management measure. The difficulties in defining the first flush and/or the questionable nature of the practical advantage in its treatment have already been discussed in detail in Section 3.2.3D and Section 6.1.1.

Shaheen (1975) has argued that even without the removal of pollutants on roadways, it could be possible to effect considerable improvements to the quality of receiving waters by altering the kinetics of pollutant transport. The changes should seek to reducing peak flows and attenuation of the flood hydrograph, thereby minimising the shock load to receiving waters. Suggested measures include, porous pavements and special kerb and gutter design. Barrett et al. (1998) have advocated the use of grass swales instead of kerb and gutter arrangement along roadways.

As discussed in Chapter 4 and 5 above, atmospheric sources too can play a significant role in contributing pollutants. Table 10 below shows the outcomes of the study by Latimer et al. (1990) on hydrocarbon concentrations in various source areas to illustrate the considerable contribution by atmospheric sources.

Table 10 – Hydrocarbon concentration in source materials (μg/g)
(adapted from Latimer et al. 1990)

Source material	Commercial	Industrial	Highways	Residential
Street dust	157	3,490	1,680	353
Roadside soil	138	856	265	38.9
Roadside vegetation	46.1	290	45.3	40.6
Atmos. deposition	2,280	125,000	-	1,940

Therefore it can be concluded that a holistic approach is needed to safeguard the quality of receiving waters in urban areas. The effectiveness of the current piecemeal approach will only be marginal and even be counter productive. It only provides a false sense of achievement and even detracts attention from the more difficult challenges to be met to safeguard urban water quality.

6.3 Future Research Directions

It is encouraging to note that there is an increasing emphasis on the safeguarding of urban water quality. This is a paradigm shift from the sole focus in the past on quantity issues for flood mitigation and the economic cost it entails. Since the 1970s the move towards urban water quality research is apparent. However the techniques and approaches adopted are strongly rooted in quantity research undertaken in the past. This applies not only to modelling philosophies and water quality models currently available, but also to the conducting of research and the reporting of outcomes.

It is important to note the strong influence exerted by chemical and biological processes in dictating the quality characteristics of urban stormwater. Therefore the extension of concepts and processes commonly adopted for urban water quantity studies will not suffice. It is important that processes other than those of physical origin are understood and taken into consideration in research studies. The mere cataloguing of quality parameters and attempts at correlation with physical catchment characteristics may provide outcomes which could be misleading. This has been the common approach adopted in a large number of the research studies reviewed. It is strongly evident that there have been only limited attempts to investigate the correlationships between quality parameters and chemical characteristics of stormwater. Atmospheric conditions and geological characteristics of the urban area add a further complexity to the issues which influence urban water quality.

These observations would help to explain the contradictory results often reported in various research studies. Secondly, it would also help to explain the difficulties commonly encountered in attempts to understand or mathematically replicate some of the fundamental tenets which underpin urban water quality research and management strategies. This includes concepts such as:

- the role of the antecedent dry period on pollutant build-up and wash-off;
- the occurrence of the first flush at the commencement of a stormwater runoff event;
- the processes of pollutant build-up and wash-off.

Consequently, the ultimate outcome of most research studies is a strong location specific aspect. This does not meant such research studies do not have any value. Rather, studies of that nature should be undertaken with the full knowledge that the outcomes will not be readily transferable. They would satisfy the need for the development of location specific management strategies.

However in the long-term and for wider benefit, it is important to ensure the transferability of the outcomes of the research undertaken. In this context, the following areas of research are considered essential for the development of an in-depth understanding of the issues which influence urban water quality:

- The correlationships between water pollutants and the physico-chemical characteristics of rainfall and stormwater runoff.

  In terms of rainfall, currently it is only the physical characteristics which are taken into consideration to account for phenomena such as rainfall detachment and transport of pollutants. However atmospheric pollution has been identified as a significant contributor to stormwater pollution in the form of wet deposition and atmospheric wash-off. Additionally, the chemical characteristics of rainfall will influence processes such as adsorption, solubility, leaching and transformation. Similarly the chemical characteristics of stormwater runoff too can bring about similar changes to pollutant wash off process kinetics.

- Study of the process kinetics of pollutant build-up and wash-off.

  This needs to be undertaken for individual pollutants as the inherent chemical processes differ. The pollutants of importance are, suspended solids, nutrients, heavy metals, hydrocarbons and organic matter.

- The role of suspended solids in urban stormwater quality.

  This is crucial as suspended solids are not only an important pollutant in their own right, but also significantly influence the process kinetics of build-up and wash-off and the bioavailability of other pollutants such as heavy metals and hydrocarbons.

- The role of dissolved organic carbon in urban stormwater quality.

  The reasons are the same as for suspended solids.

- The role of atmospheric pollutants in stormwater runoff quality.
  Atmospheric processes contribute to pollutant build-up in the form of dry and wet deposition and play a significant role in the transport of pollutants from their point of generation. Additionally, atmospheric wash-off is a significant contributor of some pollutants such as nitrogen and suspended solids to stormwater runoff.

It is important that the relationships derived are sufficiently generic to facilitate their transferability. Secondly, the research methodologies commonly used in other disciplines such as agriculture should be adapted. Large-scale catchment studies are important for relationship calibration and validation purposes. However due to the heterogeneity of large urban areas, they are not particularly suitable for the development of fundamental concepts and relationships. The use of small test plots to ensure homogeneity and tools such as artificial rainfall simulators will help to reduce the large number of variables which are usually inherent to urban water quality research.

7. CONCLUSIONS

Urbanisation can lead to significant water quantity and quality impacts. The quantity impacts of urbanisation have been well documented in research literature due to it being the primary focus of research in the past. Research into quality impacts is relatively of more recent origin. The research outcomes are often conflicting, and consequently, it is difficult to discern cause-effect relationships for urban water quality impacts. There are significant difficulties in defining the process kinetics of pollutant generation, transmission and dispersion.

Even though the sources and pathways of urban water pollutants are widely known, the current state of knowledge with regards to pollutant build-up on paved surfaces and wash-off due to rainfall events is extremely limited. The data sets available and the analysis undertaken are of little value due to their location specific nature, which preclude the identification of the range of factors which influence these processes. Anthropogenic activities and climatic and catchment characteristics together adds to the complexity of the inherent issues. Consequently, most of the fundamental concepts which underpin urban water quality modelling is contentious. This includes the role of the antecedent dry period in pollutant build-up and the occurrence of the first flush during pollutant wash-off due to a rainfall event. Also the importance of the first flush as a water quality management measure and its definition is also in question.

The primary water pollutants identified in research literature include, litter, suspended solids, nutrients, heavy metals, hydrocarbons and organic carbon. The behaviour of various pollutants and the processes they undergo are well described qualitatively in research literature. However the quantitative assessment of these pollutants is subject to significant error and the mathematical formulation of the inherent processes has not been an easy task. This is a major failure in research studies. Consequently, attempts to correlate land use to pollutant loadings have been far from conclusive. Once again, even though qualitative relationships are generally evident, the derivation of statistically significant relationships has not been satisfactory.

Therefore due to the quantitatively inconclusive nature of research outcomes, the management of water quality impacts in urban areas has proven to be an extremely challenging task. The effectiveness of commonly adopted management or structural measures such as the treatment of the first flush and street cleaning and the use of gross pollutant traps, sediment traps and detention basins is open to question. The contradictory findings in relation to various management measures clearly point to the significant role played by catchment specific factors in influencing water quality characteristics. Therefore any treatment measure adopted should be designed taking these characteristics into consideration.

However considering the fact that atmospheric sources too can play a significant role in contributing pollutants, a holistic approach is needed to safeguard the quality of receiving waters in urban areas. The current approach to urban water quality management is piecemeal and the benefits are only be marginal and even be counter productive. It provides a false sense of achievement and even detracts attention from the more difficult challenges to be met to safeguard urban water quality.

A significant drawback in current urban water quality research is that the techniques and approaches adopted are strongly based in water quantity research undertaken in the past.

The extension of concepts and processes from water quantity studies is not satisfactory due to the overwhelming reliance on physical factors and limited recognition of chemical processes. Chemical processes exert a strong influence on the quality characteristics of urban stormwater. It is this neglect which can be attributed to the often contradictory results reported in research studies and the strongly location specific nature of outcomes.

Consequently, the proposed future research directions have been formulated taking the above noted concerns into consideration. It is important to ensure the transferability of research outcomes for wider benefit and the relationships derived should facilitate this transfer.

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